A new classification approach:

Improving the Regional Ecosystem Classification System in Queensland, Australia



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Nature of Assistance	Contribution	Names, Titles and Affiliations of Co-Contributors
Intellectual support	**Collaboration**	My supervisors Associate Professor Susan Laurance and Professor Darren Crayn collaborated throughout the duration of the PhD. Other collaborators are mentioned at the start of the individual chapters where appropriate
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	Cartography and GIS	Jack Kelley, Queensland Herbarium wrote R scripts for Chapter 3

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Abstract

Dividing the world around us into categories to form classification systems is one of the fundamental tools humans use for understanding and managing the natural world. Landscape and vegetation classification systems are among the primary tools for managing the natural world and in an era of globalisation, where the landscapes that need managing cross regional and national boundaries, a standard classification system which crosses these boundaries is highly desirable.

In response to this need, in the state of Queensland in north-eastern Australia the government introduced a state-wide landscape classification system and mapping program using the Regional Ecosystem (RE) approach. This is a three-tiered hierarchy using the biogeographical classification system of Australia as the first tier. The second tier is a classification system based on geology and geomorphology and divides the landscape into broad geological groupings. The third tier is a vegetation classification system describing plant communities at the association level. The intersection of these three classification systems form a Regional Ecosystem which is defined as a plant community or communities which consistently occur on a particular substrate within a bioregion.

Casting the RE system into a global framework which compares classification systems across administrative boundaries gives an understanding of where the RE system conforms with best practice for classification systems. The classification approach of the RE system aligns with best practice by outlining the concepts and criteria for identifying communities, however, the methods for identifying these communities are reliant on supervised class definition procedures. These procedures involve the ecologist using available data combined with their own ecological knowledge and assumptions about the drivers of landscape patterns to manually identify plant communities. Although this supervised technique is common, it is not considered best practice as it is not robustly repeatable or consistent and does not produce robust

statistical information. This thesis determines, tests, evaluates and applies a suite of class definition procedures based on quantitative analysis techniques which are consistent with the concepts and criteria of the RE system. These, in combination with those concepts and criteria will, I propose, form a new classification approach for the RE system. This work took place in the non-rainforest vegetation across three different landscapes of the Cape York Peninsula bioregion in the north-east of Australia.

For class definition procedures to be adequate in identifying plant communities in a landscape, it is important to understand how well the underlying data samples that landscape. Consequently, I test how well the sampling design used to collect vegetation data for the RE system captures the environmental variability and the community and species richness within two landscapes in the bioregion. The sampling underlying the RE system is preferential, in which the location of detailed vegetation survey plots is determined by the ecologist as being representative of the surrounding community. To do this they use many qualitative data records collected during traverses across the landscape as well as patterns delineated on aerial photos. I test how well both the qualitative data records and the detailed survey plots sampled the environmental variability, using those factors expected to limit plant growth. To test the level of capture of beta-diversity and species richness I use only the detailed survey plots. The preferential sampling design underpinning the RE system sampled 98 – 100% of the environmental variability of both landscapes, a comprehensive sampling coverage. The design comprehensively captured the beta-diversity but did not adequately sample the species richness of the landscapes. This means the survey design will capture the diversity of communities in a landscape but not the floristic variability within those communities.

With an understanding of how well sampled the landscape variability and community richness is sampled, I next determine a suite of quantitative based class definition procedures appropriate for the RE system using a combination of literature review and

quantitative analysis. I found that the primary attributes needed to be based on %cover and to incorporate vegetation structure by multiplying %cover by the height of the vegetation layer. Additionally, I found that the concept of dominance varied between structural formations, with subsets of species being able to describe landscape scale vegetation patterns better than using all species. I used these findings to recommend a suite of quantitative class-definition procedures to the Queensland Government. With minor amendments they were accepted, and I use them throughout the rest of the thesis. Determining class definition procedures consistent with existing concepts and criteria is rarely done as new classification exercises using existing data are generally carried out with a new classification approach

Having determined appropriate quantitative class definition procedures, I then test them by identifying the plant communities on two landscapes in the bioregion. As is usual with classification outcomes, my new communities were assessed using a peer-review process. During this process I formalised quantitative techniques for evaluating plant communities to be used during these assessments in the future. The combination of the quantitative class definition procedures, the quantitative evaluation techniques and the peer-review process form the full suite of class definition procedures making up a new classification approach for the RE system. The new classification approach resulted in a large decrease in the number of plant communities compared with those previously identified using supervised techniques. One function of applying my new approach is to test the assumptions used by ecologists to identify communities. My results indicate incongruence in the species used to identify communities between quantitative based procedures and those used by ecologists.

To determine the differences between the communities identified using the new and the previous supervised approach I evaluated them using quantitative comparisons. This was only possible as the supervised approach used the same concepts and criteria as the RE system. I found the communities identified using the new

classification approach were more recognisable and useful for planning purposes. This was because the new approach consistently applied thresholds of dominance and consistently determined landscape scale vegetation patterns across areas with broad environmental gradients, which my results showed ecologists do not.

To establish the robustness of the new classification approach I applied it to another landscape in the bioregion to provide baseline conservation information. I chose the inter-tidal communities, as they provide important ecosystem services such as carbon sequestration but are also vulnerable to dieback from extreme climate events. Using the communities identified by my new classification approach and the accompanying RE mapping, I estimated both their potential C storage and sequestration and vulnerability to dieback from extreme El-Nino episodes. The estimated C stored in the intertidal communities was ~92% of the Australian C emissions in a year, greater than estimates for the rainforests of the bioregion which cover 3.4 times the area. The most widespread woodlands in the bioregion, which cover 16 times the area, store an estimated 1.5 times the amount of C of the inter tidal communities. Annual C sequestration potential was 0.18 – 0.34 Tg C / yr, valued between AU$8.9 - $17 million. The mangrove forests of the bioregion are among the most species diverse in the world and constitute between ~1.1 and 2.2% of the global mangrove forest. There were three mangrove forest communities vulnerable to dieback, and they were as vulnerable to dieback as those previously reported in the adjacent bioregion. Although the inter-tidal communities of the bioregion are intact, my work shows they are vulnerable to diffuse threatening processes resulting from anthropogenic change.

By determining class definition procedures consistent with pre-defined concepts and criteria of a classification system, the results of this thesis provide a deeper understanding of issues surrounding vegetation classification systems and implications of the approaches used to identify plant communities. This thesis develops a new classification approach for identifying the plant communities within the Regional

Ecosystem classification system used by the Queensland Government as a state-wide standard, thereby fundamentally changing the way Regional Ecosystems are identified across the state. This new classification approach makes the RE system more statistically robust and defensible and brings it more in line with global best practice.

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Chapter 1 General Introduction

To understand the world around us, the human brain has evolved to discover patterns in the seeming chaos (Kahneman, 2011). What those patterns are, how to group by similarity or separate by dissimilarity is the essence of classification. Vegetation classification aims to provide a framework for ordering, describing and understanding the patterns observed in the vegetation mantle covering the landscape (Whittaker, 1973b). Based on the underlying assumption that patterns of species are repeated, identifying these patterns through a classification exercise allows us to understand the connections and similarities between plant communities and landscapes across varying geographical areas (Whittaker, 1973b), thus providing baseline data for contextualising information in the vegetation mantle (Peet & Roberts, 2013).

Vegetation classification systems form a base for land management and the ecological exploration of the patterns and drivers of species' distributions (K. R. Clarke & Warwick, 2001; Kent, 2012). Applying a vegetation classification system for management purposes through a map showing geographical areas of similarity within the system is common and maps are therefore an oft associated component. Vegetation classification systems and accompanying maps require a simplification of the complexity of the natural world. A classification system may describe the full floristic composition of areas, however a map which describes this detailed composition quickly becomes too complicated for practical use. Contrastingly, a map which does not describe the complexity enough is inadequate for management of the areas depicted (Kuchler, 1951). Hence a vegetation classification system and an accompanying map are interdependent. A vegetation classification system and accompanying map that combines physiognomic, floristic and ecological approaches to describing the complexity of nature is most useful for land management, as it describes information useful for a variety of purposes and provides it an easily accessible fashion (Federal

Geographic Data Vegetation Subcomittee, 2008; Kuchler, 1951). The demand for vegetation classification systems and maps is steadily increasing because of their direct applicability across a broad range of issues (Chytrý, Schaminee, & Schwabe, 2011; Wesche & von Wehrden, 2011) and they are often used as a surrogate for measuring biological diversity (Peet & Roberts, 2013) underpinning many land management decisions and much scientific research (Chytrý et al., 2011; De Cáceres & Wiser, 2012; Jennings, Faber-Langendoen, Loucks, Peet, & Roberts, 2009). Accompanying maps specifically allow interrogation of changes in extent as well as composition (Accad, Neldner, Kelley, Li, & Richter, 2019; Küchler & Zonneveld, 1988; Mucina & Daniel, 2013). Reflecting these uses, vegetation classification systems and maps are increasingly tied to legislation at international, national and regional levels (European Commission, 2003; Queensland Government, 1999).

The globalisation of planning and management issues have created an increasing need to manage landscapes across geographical and administrative boundaries (Peet & Roberts, 2013). To do this, it is desirable to have a consistent vegetation classification system crossing these boundaries (De Cáceres et al., 2015; S. Franklin, 2015). Recognising this, the government of the state of Queensland in north eastern Australia, adopted a state-wide landscape classification system in 1999 (Sattler & Williams, 1999). The state covers an area of 1.7 million km^2 and has a sparse population of 4.6 million people. Approximately 80% of its area is natural vegetation, of which 98.5% is sclerophyll and 1.5% is tropical forest (Accad, Neldner, et al., 2019). The government adopted the Regional Ecosystem (RE) classification system which is a hierarchical landscape classification system with vegetation as its lowest level (Figure 1.1). The hierarchy is three-tiered with the first division being based on the Interim Biogeographical Regions of Australia (Thackway & Cresswell, 1995). The second division of the hierarchy is termed ‘land zone’; a concept that involves broad geological divisions of the landscape with consideration of geomorphological processes and soils

(Wilson & Taylor, 2012). Examples of land zones include ‘alluvial river and creek flats’, ‘coastal dunes’ or ‘hills and lowlands on granitic rocks’. The third level of the classification scheme is termed ‘vegetation community’ and consists of plant communities identified at the plant association level. An RE is therefore defined as “a vegetation community, or communities, in a bioregion that are consistently associated with a particular combination of geology, landform and soil” (Sattler & Williams, 1999), noting that an RE may contain more than one vegetation community, but a vegetation community cannot occur in more than one RE. Thus, the RE classification system (the RE system) incorporates geodiversity, as well as floristic diversity. REs are revised periodically as new data are supplied and, to this end, each bioregion has a technical committee whose role it is to review and implement proposed changes based on appropriate data. This technical review committee performs the same function as similar panels in international and other Australian jurisdictions (European Vegetation Survey Working Group, 2017; Federal Geographic Data Vegetation Subcomittee, 2008; Office of Environment & Heritage & NSW Office of Environment and Heritage, 2018). Recognising the importance of maps in the role of management, a Government funded state-wide RE mapping program commenced in 1999 with the introduction of the RE system. REs are mappable entities with a distinctive signature recognisable from remote sensing imagery at the landscape scale of 1:100,000. REs form the basis for mapping and survey projects at all scales across the state and are embedded in both national and state government legislation (Department of Agriculture Water and the Environment, 2009; Queensland Government, 1999). They have become the fundamental baseline dataset for biodiversity information across the State.

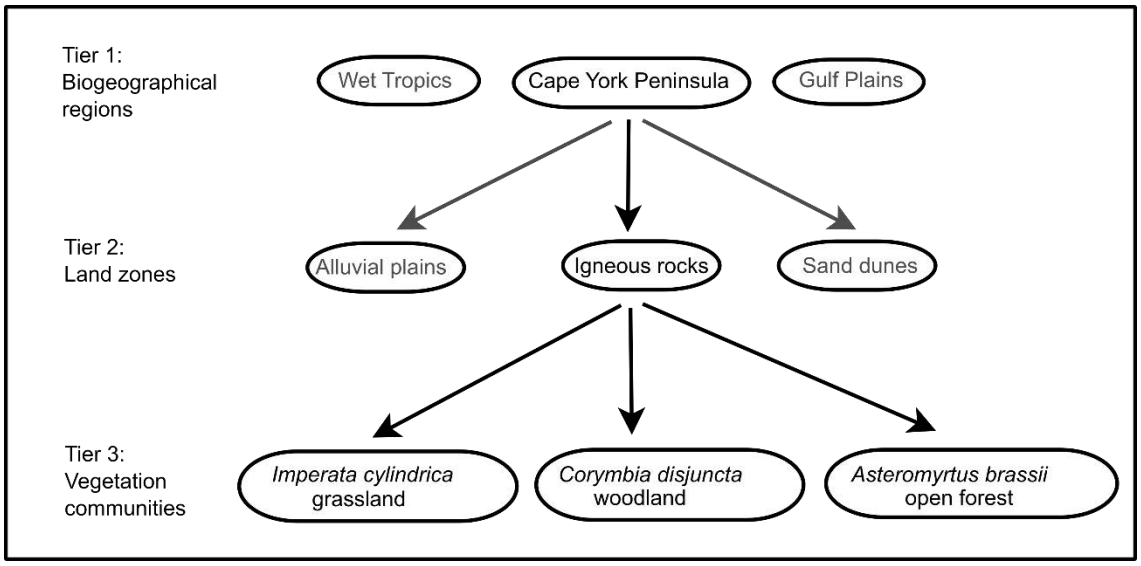

Figure 1.1: Regional ecosystem classification system. Regional ecosystems are a three-tiered hierarchy. The first tier is biogeographical regions based on the Interim Biogeographical Regions of Australia. The second tier is broad geological / geomorphological groups (labelled land zones). The third tier are plant communities recognised at the association level (labelled vegetation communities).

Many scientific administrations around the world have recognised a need for a standardised vegetation classification system that crosses local, state and even national boundaries as well as spanning multiple environmental regions (Peet et al., 2018; Rodwell, 2018; Walker et al., 2018). Most administrations that have developed boundary-crossing classification systems have had, as their starting point, multiple existing classification systems and maps developed for small geographic areas, for specific purposes and with a classification system developed in isolation to surrounding areas (L. R. Brown & Bredenkamp, 2018; Faber-Langendoen, Aaseng, Hop, Lew-Smith, & Drake, 2007; Mucina et al., 2016; Rodwell, 2018). Classification systems and maps for geographic areas adjoining each other are often not relate-able (Küchler & Zonneveld, 1988). Administrations have dealt with this situation in different ways. In the United Kingdom and the United States of America, new umbrella classification systems to which all new classification exercises must relate have been imposed from the top down (Federal Geographic Data Vegetation Subcomittee, 2008; Rodwell, 2006). Across continental Europe the situation differs. Because of the long history of

vegetation classification across the continent there are many classification systems at the plant association level with detailed information and data to support them (Mucina et al., 2016). In this case, scientists have worked to relate these many systems to each other to produce a European Vegetation Classification (Mucina et al., 2016), essentially forming a classification system starting at the lowest level of the hierarchy and moving upwards. However, there is still a need to unify classification protocols across the continent (Marcenò et al., 2018). In New Zealand, where there was an existing national classification (Wiser, Hurst, Wright, & Allen, 2011), vegetation scientists have modified and extended this, keeping old units and developing new ones thus melding old and new systems (Wiser & De Cáceres, 2013). However, this new system was developed using a different approach so old classification systems are not directly relatable to the new one. Across South Africa consistent classification protocols are used, but there is no formal hierarchical classification system (L. R. Brown & Bredenkamp, 2018). The situation in Queensland contrasts with these examples in having a limited number of fine scale classification systems covering small geographic areas. Rather, large parts of the State were described at the broad classification landscape level with vegetation types described using expert knowledge (for example reports included in the Western Arid Land Use Study https://publications.qld.gov.au/dataset/land-systems-warlus-fwa2 accessed 29/8/19 and the Land Research Surveys http://www.publish.csiro.au/CR/issue/5812 accessed 29/8/19). These vegetation types were used as a basis for, and integrated in to, the RE system with the plant communities comprising REs also identified using an expert-based (supervised) classification approach.

Classifying vegetation patterns into vegetation types has a long history (Goodall, 2014) with a consequent evolution of ideas, concepts and methods (Peet & Roberts, 2013). The vegetation types recognised from any classification exercise are largely dependent on the purpose and scale of the final classification system (Gillison, 2012) and

consequently there is a plethora of classification systems emphasising different attributes such as floristic composition, functional traits, dominance, structural composition or combinations of these (Peet & Roberts, 2013; Whittaker, 1973b). Many of these classification systems were developed with system-specific terminologies describing the processes and components of each. However, with globalisation has come the need to relate systems developed in isolation to each other (De Cáceres et al., 2015). To this end a framework and terminology for comparing vegetation classification systems and the processes used to develop them has recently been proposed (De Cáceres et al., 2015, 2018). In this framework plot-based classification of vegetation is broken into two distinct sections, comprising the structural elements and the procedural elements respectively. The structural elements include the vegetation plot data, the vegetation type identified by the classification exercise and the classification system itself (made up of vegetation types). The primary procedural element is the classification approach. This includes the concepts and the classification protocols used to define vegetation types. The classification protocols, in turn include the criteria and the class-definition procedures used to identify the vegetation types. These procedures include such elements as the data collection methods, taxonomic resolution, the primary vegetation attributes and the plot-grouping techniques. Primary vegetation attributes are those attributes of the vegetation specifically used to consistently group plots into vegetation types (for example species, abundance or physiognomy). Any associated environmental attributes used to help to align plots to vegetation types are considered as secondary attributes. Plot-grouping techniques are also cast into a consistent terminology (De Cáceres et al., 2015). Those based on expert-knowledge and manual grouping of plots are termed supervised, those that incorporate expert-based and quantitative methods are termed semi-supervised and quantitative techniques with no input from experts are termed un-supervised. This interpretation of these terms differ from those in De Cáceres & Wiser (2012) who adopted the machine learning interpretation of ‘supervised’ as labelled training data.

The structural and some procedural elements of the RE system are defined in accompanying documentation outlining standardised survey and mapping methods (the Queensland Methodology) (Neldner, Wilson, et al., 2019). Comparing these with international vegetation classification systems included in a special issue of the journal Phytocoenologia (volume 48, issue 2 (2018) De Cáceres et al., 2018 Table 1) allows an understanding of the similarities and differences of the RE system with those used elsewhere. The RE system, along with all the systems included in the special issue, has the plant association as the lowest classification level (this is a reflection of scale and not importance). Most systems place the plant association within a hierarchy of classification levels based on vegetation characteristics (for example; alliance and formation). Contrastingly, the RE system appears to be unique in formally including the environmental variables of biogeographical and geological divisions of the landscape as mandatory structural elements of the classification system. The plant communities making up REs are however, also used in a different conceptual hierarchy to form the Broad Vegetation Groups of Queensland, which more closely align to the concepts of alliance and formation levels used in other classification systems (Neldner, Niehus, et al., 2019). The only other classification system reviewed which used a low classification level to form another conceptual hierarchy was the Biogeoclimatic Ecosystem Classification used in Canada (MacKenzie & Meidinger, 2018). In line with other systems in countries where managing existing natural vegetation types is the primary purpose of the system, the ecological scope of the RE system is confined to natural vegetation, (L. R. Brown & Bredenkamp, 2018; MacKenzie & Meidinger, 2018; Walker et al., 2018; Wiser & De Cáceres, 2018). Countries where highly modified landscapes predominate all include semi-natural and cultural vegetation types in their classification system (Federal Geographic Data Vegetation Subcomittee, 2008; Gillet & Julve, 2018; Guarino, Willner, Pignatti, Attorre, & Loidi, 2018; Rodwell, 2018).

The RE system has many procedural elements in common with those systems reviewed. Although it uses one set of environmental variables as structural elements, it also uses as procedural elements environmental attributes similar to other systems (De Cáceres et al., 2018). Like most, the RE system is plot-based (the exception to this is the system used in China, (Guo et al., 2018)). However, like the Chinese system, the primary attributes used to define communities in the RE system are dominant species in vegetation layers. As well as incorporating a species dominance approach the RE system incorporates a physiognomic approach to identifying plant communities by using vegetation structure as an identification criterion. Although identifying communities based on dominance was more common in the past (Whittaker, 1973a) it is not unusual today especially in landscapes of low species richness (Faber-Langendoen et al., 2014; Landucci, Tichý, Šumberová, & Chytrý, 2015; Wesche & von Wehrden, 2011). Plot-based identification of communities using vegetation structure, however, is less frequent, as most systems using this approach are not plot-based (De Cáceres et al., 2015). Rather than using dominant species, all other systems reviewed use the full species composition of vascular plants as their primary attribute of classification (De Cáceres et al., 2018). The RE system specifies a standard plot size of 500 m^2, shown to adequately capture the alpha diversity of plots in non-rainforest vegetation in Queensland (Neldner & Butler, 2008). This contrasts with all other systems reviewed, which have variable plot sizes. Also, in contrast to all but the Chinese system, plot-grouping techniques identifying communities are fully supervised in the RE system. All others either already incorporate, or are working to incorporate (Faber-Langendoen et al., 2014) un-supervised or semi-supervised plot-grouping techniques to identify vegetation types at the lowest level of the classification hierarchy. Consequently, unlike most of the systems reviewed, there is no evaluation of the effectiveness of communities within the RE system using the characteristics of the communities themselves (internal evaluation). There is only external evaluation through

peer-review. REs are therefore currently identified and evaluated using fully supervised plot-grouping techniques.

Supervised plot-grouping techniques to identify communities are most often used in remote areas with limited researchers such as in Queensland (Peet & Roberts, 2013). However, these techniques have acknowledged problems including their lack of transparency, repeatability and consistency between researchers (Kent, 2012; Mucina, 1997; Oliver, Broese, Dillon, Sivertsen, & McNellie, 2012). The outcomes from such processes are heavily dependent on a researcher's knowledge of the vegetation of the area and are also biased by a researcher's assumptions of the ecological and biophysical processes important to landscape function and biodiversity (Kent, 2012). Consequently, supervised methods do not produce communities that are statistically comparable (Harris & Kitchener, 2005; Kent, 2012; Oliver et al., 2012). Using un-supervised plot-grouping and evaluation techniques in a classification approach can help to overcome some of these problems and is regarded as global best practice (De Cáceres et al., 2018; Kent, 2012; Peet & Roberts, 2013). It is imperative that the classification approach of the RE system follow global best practice, as the management decisions made using this system affect people's livelihoods, the biodiversity of the State and the future management of ecosystems across a vast area and in an era of unprecedented impacts due to climate change (IPCC, 2014).

The RE system's classification approach as outlined in the Queensland Methodology (Neldner, Wilson, et al., 2019) has criteria specifying that communities are identified at the plant association level using plot-based records and the pre-dominant layer defined as that contributing most to the above-ground biomass (Neldner, Wilson, et al., 2019). Communities are defined using the height, cover and dominant species in this pre-dominant layer, with sub-ordinate consideration given to associated species in other layers. Plant associations are thus defined as a community where the pre-dominant layer has a uniform floristic composition and exhibits a uniform structure (Neldner,

Wilson, et al., 2019) aligning with both the Beadle (1981) definition of a plant association and a necessary emphasis on canopy species used for vegetation mapping (Appendix 1.1). These concepts and criteria form the scaffolding for choosing appropriate un-supervised plot-grouping and internal evaluation techniques for a new quantitative based classification approach within the RE system. Although the Queensland Methodology (Neldner, Wilson, et al., 2019) recommends using un-supervised plot grouping techniques as part of its classification approach it gives no guidance on doing this.

Overview of the Thesis

This doctoral thesis addresses this need to develop a new quantitative based classification approach, thus moving it from a fully supervised to a semi-supervised approach.

The thesis contains seven chapters: the introduction (this chapter), five data chapters and a final synthesis chapter. To avoid repetition, I have included a description of the study area and the data collation in this introduction chapter as many of the chapters use the same study area and data. Consequently, I have removed these sections from the chapters which are published or prepared as journal articles. The five data chapters are based on a review of the literature and quantitative analysis of empirical data. Chapters 2, 4 and 5 have been published as journal articles. Chapter 3 includes a published manuscript and Chapter 6 has been accepted as a journal article. Although I have removed most of the repetition between chapter publications, there will still be some, primarily in the introduction and method sections. Chapters 2 and 4, treated separately in this thesis, were combined in one publication.

Interpreting and applying a classification system requires an understanding of the biases in the dataset. Therefore, in my first chapter I assess how well the vegetation sampling design used by the Queensland government, samples the environmental variability, the beta-diversity and species richness across the landscape. I then develop

a new classification approach for the RE system by determining appropriate class-definition procedures. These include un-supervised plot-grouping and evaluation techniques which are conceptually consistent with the established concepts and criteria. I test my new approach by identifying the plant communities using Cape York Peninsula biogeographic region as a case study. Because both the existing and new classification approach use the same concepts and criteria, I quantitatively evaluate the differences in plant communities identified by these two approaches. Finally, as vegetation classification systems are ultimately a tool, I apply my new classification approach to another landscape in an ecological context. I identify the inter-tidal communities within the bioregion and use this to provide baseline ecological information important for their conservation management. As each of these investigations was a separate component of the overall project this thesis is structured as a series of stand-alone publications.

Summary of chapter 2: Assessing the vegetation survey design adopted by the Queensland government

To interpret and apply a classification system it is important to understand the biases in the survey design which underpins it. The survey design adopted by the Queensland government is preferential in that sampling plots are located at sites considered to represent the plant community of the surrounding area. In this chapter I evaluate this preferential sampling design, assessing how well it captures the environmental variability, the beta-diversity and the species richness in two landscapes in my study area. This also allows an evidence-based answer to a question frequently asked by users of the RE system, “how adequate is the sampling”? This chapter is based on the publication Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia,* 18, 29–71, with revisions.

Summary of chapter 3: Determining appropriate class definition procedures to form a new classification approach in the RE system

In this chapter I determine and recommend appropriate un-supervised techniques to incorporate into the class definition procedures for a new classification approach for the RE system. I use a combination of, a consideration of the existing concepts and criteria, a review of the literature and empirical analysis. The latter is published as Addicott, E., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology,* 19, 67–76. These recommendations were considered at a Queensland government sponsored workshop comprised of vegetation mapping practitioners and experts on the vegetation of northern Queensland. With few amendments they were adopted as Queensland government practice, forming the new un-supervised class definition procedures to identify plant communities within the RE system. These procedures have been adopted in all subsequent chapters.

Summary of chapter 4: A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion

Following on from determining new class definition procedures, I test them by identifying and describing the plant communities on two landscapes, covering ~53 000 km^2 of Cape York Peninsula bioregion. Through this I formalise a full suite of class definition procedures including external evaluation techniques. The plant communities identified in this chapter form the revised Regional Ecosystems for these landscapes and were incorporated into the state-wide Regional Ecosystem mapping program (v11 available on line at http://qldspatial.information.qld.gov.au/catalogue/custom/index.page). In this chapter, I also produced characterising species for plant communities using statistical

techniques, hitherto not done within the RE system. This chapter is published in the same publication as chapter 2 (Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia,* 18, 29–71). The suite of techniques used in this chapter form the new classification approach for the RE system.

Summary of chapter 5: Supervised versus un-supervised classification: A quantitative comparison of plant communities in savanna vegetation

An obvious question to ask after applying new class definition procedures to an area is "what are the differences between the old and the new plant communities identified"? In this chapter I quantitatively assess these differences (published in Addicott, E., & Laurance, S. G. W. (2019). Supervised versus un-supervised classification: A quantitative comparison of plant communities in savanna vegetation. *Applied Vegetation Science,* 22, 373-382). To do this, I used evaluation criteria based on the recognisability of communities and their usefulness for land management purposes. Quantifying the differences between communities identified by these two approaches is rare as communities within the same study area are generally identified by approaches differing in their classification concepts and criteria as well as the plot-grouping techniques.

Summary of chapter 6: Applying the new classification approach in an ecological context: the inter-tidal plant communities in north-eastern Australia, their relative role in carbon sequestration and vulnerability to extreme climate events

One of the functions of a classification system is as a tool for further exploration of the landscape. In this chapter, I demonstrate the applicability of my new classification approach in an ecological context using the inter-tidal communities. These communities are recognised globally as providing important ecosystem services such

as carbon storage and sequestration, in mitigating the impacts of climate change, and are extensive along the 7,400 km Cape York Peninsula bioregion coastline. However, during the strong El-Nino event of 2015-16, large-scale dieback of mangrove forests was observed in the Gulf of Carpentaria region adjacent to Cape York Peninsula bioregion. This chapter demonstrates the applicability of my new classification approach by providing baseline information on the floristic composition, the ecosystem services of carbon storage and sequestration, and vulnerability to climate extremes of these communities to underpin their effective conservation management. Please note this chapter is in review with *Aquatic Conservation: Freshwater and Marine Ecosystems* and is written in the third person to comply with the journal formatting requirements.

Summary of chapter 7: Synthesis and discussion

In this chapter I review and discuss my results in a global context and assess issues around implementing my new approach and future research directions.

Study area

Cape York Peninsula bioregion (CYP) covers 120 000 km^2 in the monsoon tropics of north-eastern Australia and lies between 10 and 16 degrees south (Figure 1.2). Elevations range from sea level to approximately 800 m. The annual average rainfall varies between 1000– 2000 mm with 80% falling in the wet season between December and March (Horn, 1995). Temperatures range from an average annual monthly minimum of 14 ^{o}C in winter (July) to an average monthly maximum of 35 ^{o}C in summer (December) (Bureau of Meteorology, 2016). My research encompasses the savanna and inter-tidal plant communities on three of the ten land zones occurring on CYP (Neldner, 1999); the old loamy and sandy plains (land zone 5), the ranges and hills on igneous rocks (land zone 12) and the inter-tidal communities (land zone 1) (Figure 1.2). The old loamy and sandy plains (land zone 5) are remnant Tertiary formations of CYP

and are distributed across the full extent of the bioregion covering 48 000 km^2 (40% of the bioregion). In this thesis I refer to this as the Tertiary landscape. The ranges and hills on igneous rocks (land zone 12) of CYP and occur primarily along a north-south spine associated with the Great Dividing Range and cover 6 500 km^2 (5% of the bioregion) (Figure 1.2). In this thesis I refer to this as the igneous landscape. I used these two landscapes to test the Government's preferential sampling design and to develop and test my new classification approach. The inter-tidal communities (land zone 1) are distributed around the extent of the 7 400 km CYP coastline (Figure 1.2) and I use this landscape to apply the new classification approach in an ecological context. Where more detailed characterisation of each landscape is necessary it is included in the appropriate chapters. Although the landscapes are referred to with a numbered land zone in the RE classification system, I avoid this through the body of my thesis, instead referring to the Tertiary landscape, the igneous landscape and the inter-tidal communities. The RE numbering system is maintained in the appendices.

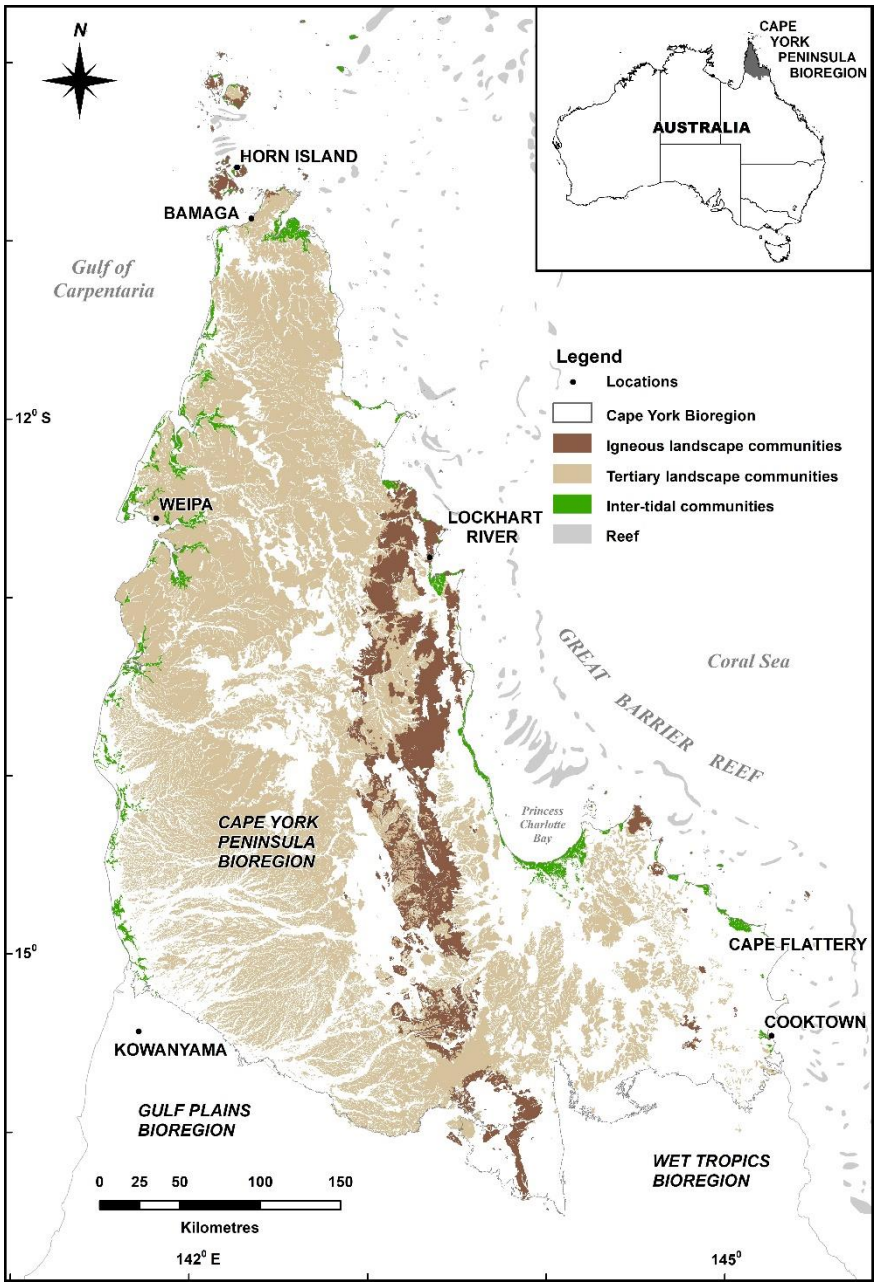

Figure 1.2 Distribution of the landscapes in Cape York Peninsula bioregion used in this thesis

Data Collation

My research took advantage of existing vegetation plot-data largely collected as part of the Cape York Peninsula Land Use Study (CYPLUS) carried out in the early 1990s (Neldner & Clarkson, 1995) with more plot data later collected as part of the RE mapping program. All plot data were collected in accordance with the Queensland Methodology (Neldner, Wilson, et al., 2019). There are three types of plot data

collected at different levels of detail. The first contains detailed quantitative vegetation data and referred to as ‘detailed’ plots throughout the rest of the thesis. This methodology requires a full floristic survey of a 500 m^2 plot, with percent foliage projected cover (%FPC) for each species, in each woody vegetation layer, recorded along a 50m transect using the line intercept method as well as stem counts and basal area for each species in each woody layer in the plot. The average height of each layer was also recorded. The ground layer had species abundance recorded as an estimate of %FPC in 1 m^2 quadrats at 10 m intervals along the 50 m transect (five quadrats in total) and averaged. It is only these detailed plots which are used in quantitative analyses throughout my thesis, unless otherwise stated. The second were plots containing quantitative data that was incomplete or collected using different methods and are referred to as ‘non-detailed’ plots throughout. The third were observational records which contain qualitative data and were rapidly collected in large numbers during field traverses of the mapping area. They included a record of geolocation, dominant species in the pre-dominant layer and vegetation structure. Observational records were extracted from GIS coverages associated with the mapping project while other vegetation plot data was extracted from the Queensland government ‘CORVEG’ database. Vegetation communities in which the pre-dominant canopy was the ground layer I refer to collectively as grasslands, but this group includes sedgelands and rock pavements with scattered herbs and forbs as well as true grasslands (Neldner, Wilson, et al., 2019).

The number of observation records, plots and species used for each section of this thesis differs based on the analysis performed and is detailed in each chapter.

Chapter 2 Assessing the vegetation survey design adopted by the Queensland government

This chapter is based on a published paper (Addicott et al., 2018), but has been revised:

Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia*, 18, 29–71. doi:10.7751/cunninghamia.2018.18.003

Statement of contribution of others:

Authors contributing to this section of the paper and this chapter were Addicott, Laidlaw and Laurance. Addicott conceived the idea, conducted all analyses and wrote the manuscript. Laidlaw provided advice on analyses, provided environmental datasets and commented on the paper. Laurance provided intellectual advice and commented on the paper.

Contextual overview

To identify and describe the plant communities in the RE system the landscape is sampled using plots to collect vegetation data. The survey design for collecting data underpinning the RE system is preferential in that plots are located at a site which is representative of the plant community. Most survey designs have biases and understanding these biases is important in applying a classification system. In this chapter, I assess how well this design, adopted by the Queensland government, captures the environmental variability, the plant community heterogeneity (beta-diversity) and the species richness on the Tertiary and igneous landscapes of CYP.

Introduction

Identifying and characterising plant communities in a plot-based classification exercise is based on data collected at specified plots across the landscape ((Whittaker, 1973b). The density and distribution of those plots requires a decision about the sampling design and will be dependent on the purpose of the classification exercise (De Cáceres et al., 2015; Peet & Roberts, 2013). There are a number of possible approaches to collecting plot data to sample a study area, summarised by De Cáceres et al. (2015) as random, systematic, preferential and stratified random. There has been much debate about the advantages and disadvantages of different sampling designs used for vegetation survey (Botta-Dukát, Kovács-Láng, Rédei, Kertész, & Garadnai, 2007; Diekmann, Kühne, & Isermann, 2007) with preferential sampling producing data that has lower statistical power than the other approaches but a higher ability to sample plant community heterogeneity (beta-diversity), species richness and rarity in the landscape (Bell, 2013; Chiarucci, 2007; Roleček, Chytrý, Hájek, Lvončik, & Tichý, 2007). With preferential designs, statistical procedures are descriptive and not inferential, however there is a higher likelihood of having comprehensive coverage of the study area (Peet & Roberts, 2013).

In a vegetation survey which aims to identify and characterise the range of plant community types in an extensive study area some level of preferential placement of plots is necessary (Peet & Roberts, 2013). The heterogeneity of plant community distribution (beta-diversity) and the environmental variability of a landscape are closely linked, with greater variability of the latter generally leading to greater beta-diversity (Kent, 2012; Lepš, 2005). To maximise the capture of beta-diversity and species richness it is therefore necessary to have adequate spatial distribution of plots across major environmental gradients (Kent, 2012). If a new quantitative based classification approach is to be developed in the RE system, it is important to understand how well these three levels of diversity are sampled by the Government's standard survey design. This design is preferential in that detailed plots are located at sites which are representative of the surrounding plant community. These representative locations are chosen using the rapid observational records collected during survey traverses through all accessible areas of a landscape (Neldner, Wilson, et al., 2019). The underlying assumptions of the plot locations are that they are not ecotonal and do not cross community boundaries. In this chapter, I assess how well this two-tiered preferential design captures the environmental variability, the beta-diversity and the species richness of the two landscapes of the bioregion.

Methods

Dataset

The data included in this assessment encompassed the Tertiary and igneous landscapes of CYP (Figure 1.2) and were all plots and observational records collected at any of the three levels of detail in the standard data collection procedures. Plot data may be biased in a number of ways (Lájer, 2007; Michalcová, Lvončík, Chytrý, & Hájek, 2011; Neldner, Crossley, & Cofinas, 1995), however many potential biases in the detailed plots were reduced as data was collected using standardised plot size, data collection methods, data attributes, data quality and season of survey, with those plots dominated by the ground layer surveyed between May and August (the early dry season). Detailed plots located in

ecotones were excluded. This resulted in 192 detailed plots, 38 non-detailed plots and 4,670 observational records on the Tertiary landscape and 96 detailed plots, 45 non-detailed plots and 1,424 observational records on the igneous landscape.

Although the criteria for recognising an RE centre around dominant species, detailed plot data contains full floristic sampling. Therefore, to understand how well the survey design captures the species richness and beta-diversity of the landscape I used all species captured by the detailed plot data with the following restrictions. Taxa identified only to Family level were removed, as were taxa identified only to Genera level (unless there was only one record of that genus in the dataset) and non-native species. Taxa identified to subspecies level were amalgamated to species level. This resulted in a total of 620 taxa on the Tertiary landscape and 552 on the igneous landscape.

Analysis

Environmental variability

To assess the extent to which the sampling design captured the environmental variability of each landscape I firstly chose environmental variables expected to limit plant growth. Using these and the DOMAIN software (Carpenter, Gillison, & Winter, 1993) I determined the spatial distribution of areas environmentally similar to, firstly, all plots and observational records and then, separately, to only detailed plots. I conducted the first analysis as the observational records and non-detailed plots assist in identifying locations for the detailed plots. This analysis will give an understanding of how much of the environmental variability was surveyed in contrast to being sampled with detailed data.

In choosing environmental variables for assessment I followed the convention of testing those expected to limit plant species growth, dividing them into climate and soil themes. I used four climate variables; two temperature variables (average annual temperature, and the coefficient of variation of temperature seasonality) and two rainfall variables (annual average rainfall, and the mean moisture index of the lowest quarter), available as ANUCLIM

datasets (Xu & Hutchinson, 2013). The soils variables were grouped into soil nutrients (organic carbon content, and phosphorus) and soil structure (available plant water capacity, permeability, drainage, and slope) (Lyons, Foster, & Keith, 2017; Neldner, Laidlaw, et al., 2017). All soil datasets came from Australian Soil and Resource Information System ("Australian Soil Resource Information System," 2014; McKenzie, Jacquier, Maschmedt, Griffin, & Brough, 2012), with the slope derived from the digital elevation model for CYP (GeoScience Australia, CSIRO Land and Water, & Bureau of Meteorology, 2009). In addition to these climate and soil variables, I assessed how well the survey sampled variation in vegetation structure by using a maximum persistent greenness GIS coverage (Joint Remote Sensing Research Project, 2017). This coverage is derived from LANDSAT imagery classification and, on CYP, equates to density of woody vegetation layers, with a higher greenness index indicating denser woody vegetation. While density of woody vegetation is significantly correlated with the climate variables ($r = 0.6$, $p < 0.0001$), the R^2 value of the 4-way multiple regression is 34% indicating the predictability of density of woody vegetation using these variables is relatively low (Appendix 2.2. Investigation into the correlations between persistent greenness index and climate variables.). I am therefore confident that assessing woody vegetation density will provide useful additional information on bias in sampling of vegetation structure. All these datasets were accessed as raster coverages with 30m grid cells.

The DOMAIN software (Carpenter et al., 1993) was developed as a species distribution modelling program using pattern analysis techniques and has been successfully used in recent applications (Gillison et al., 2016). DOMAIN overlays each environmental variable to create an environmental 'envelope' for each grid cell in the raster datasets across the landscape. It then extrapolates the correlation between plot locations and the 'envelope' across the landscape using the Gower metric to quantify similarity. This metric uses range standardisation to equalise the contribution from each variable as opposed to variance standardisation which is more susceptible to bias from high densities of plot locations

(Carpenter et al., 1993). The result is a %similarity of the environmental 'envelope' at each grid cell to any plot location, with 100% similarity being maximum congruence. I divided the range of similarities into classes (>95% similarity, 95 - 90%, 89 - 75%, <75%) to display areas which range from well sampled to under sampled.

Beta-diversity

Beta-diversity, or the heterogeneity of plant communities across the landscape, can be measured by the turn-over of species between plots (Kent, 2012) and the Bray-Curtis dissimilarity index is an oft used metric for this (Lepš, 2005; Roden, Kocsis, Zuschin, & Kiessling, 2018). Using the detailed plots for each landscape and PRIMER-e v7 software package (K. R. Clarke & Gorley, 2015), I calculated the beta-diversity as the mean Bray-Curtis dissimilarity index between all pairs (with cover as abundance and square-root data transformation) (Havlová, Chytrý, & Tichý, 2004). To estimate how well the beta-diversity was captured for each landscape I calculated 90% confidence intervals using the dissimilarity matrices of 50 subsets of plots (Botta-Dukát, 2008). Each subset was a randomly chosen 100 plots (from 192) on the Tertiary landscape and 55 plots (from 96) on the igneous landscape.

Species richness

Estimating the size of species pools is an active area of research, with a number of approaches available (K. R. Clarke & Gorley, 2015; McCune & Grace, 2002; Oksanen et al., 2019). Species accumulation curves display the extent to which sampling has captured the expected number of species in an area (McCune & Grace, 2002). The curve is expected to asymptote as the capture of species approaches the full pool, and the steepness of the curve is indicative of sampling completeness (Kent, 2012). I graphed the species accumulation curves for both landscapes using the recommended Kindt's exact method in the 'vegan' R package (Oksanen et al., 2019). While species accumulation curves are useful

in graphically displaying whether the full species pool has been captured, other methods estimating the total number of species in a pool centre around estimating the number of unseen species (Oksanen et al., 2019). There are both parametric and non-parametric approaches, with non-parametric being most useful for ecologists as they do not require assumptions about the distribution of data (Chiarucci, Enright, Perry, Miller, & Lamont, 2003). The three most commonly used nonparametric approaches are the Chao estimators (Chao1 and Chao2), jacknife estimators (1st and 2nd order) and bootstrapping (Chiarucci et al., 2003; Oksanen et al., 2019). While there are arguments supporting the use of any one of these estimators, I have assessed how well the data captures species richness by comparing the number of species sampled to the estimates from four of the estimators in the 'specpool' function of the R package 'Vegan' (Oksanen et al., 2019). This allows a general comparison of the degree to which the species richness of each landscape is captured with the estimated full species pool. As the plot data measures abundance using crown cover and not counts, it is therefore appropriate to use estimators which are based on presence / absence. Specifically, I used the Chao2 estimator, the 1st and 2nd order jacknife estimators and the bootstrap estimator.

Results

Environmental variability

The survey design comprehensively sampled the full environmental variability in each landscape. Between 99 and 100% of the total area of each landscape was >90%-similar to any observational record or plot for all variables (Table 2.1). Results were similar for detailed plots for climate, vegetation structure and soil nutrient variables. Between 99 and 100% of the total area of Tertiary landscape and 98% of the igneous landscape was >90%-similar to any detailed plot (in the respective landscape). These results were slightly lower for soil structure, with 98.6% of Tertiary landscape and 95% of the igneous landscape, >90%-similar to any site (Table 2.1). Observational records and non-detailed plots surveyed more environmental variability than detailed plots. When the area surveyed that was >95% similar

is considered, the differences become apparent, ranging between 9.1% (for soil structure) and 0.78% (for vegetation density) on the Tertiary landscape, and 19% (for climate) and 2.7% (for soil structure) on the igneous landscape (Table 2.1). Appendix 2.1 has indicative maps of areas of <90% similarity to plots. The detailed GIS coverages of these areas are available from the senior author if more detail is required.

Table 2.1 Total area of Tertiary and igneous landscapes at different similarity levels to any observational record or plot for each environmental variable. For example, 8 km^2 of the Tertiary landscape is between 75 – 89% similar in climate to any observational record. This represents 0.01% of the total area of the landscape. The minimum similarity in climate of any grid cell to any observational record is 81%. Figures are rounded to the nearest km^2.

Tertiary landscape	Observational records				Detailed plots			
	% Similarity Class	km^2	**% total area**	Minimum %similarity	% Similarity Class	km^2	**% total area**	Minimum %similarity
Climate	<75%	0	**0%**	81	<75%	2	**0.003%**	70
	75 - 89%	8	**0.01%**		75 - 89%	261	**0.41%**	
	90-95%	33	**0.1%**		90-95%	4006	**6.32%**	
	>95%	63339	**99.9%**		>95%	59111	**93.26%**	
Vegetation Density	<75%	2	**0.004%**	63	<75%	37	**0.06%**	12
	75 - 89%	7	**0.01%**		75 - 89%	64	**0.10%**	
	90-95%	7	**0.01%**		90-95%	407	**0.64%**	
	>95%	63363	**99.98%**		>95%	62873	**99.2%**	
Soil Nutrient	<75%	1	**0.001%**	66	<75%	22	**0.04%**	0
	75 - 89%	1	**0.001%**		75 - 89%	3	**0.004%**	
	90-95%	2	**0.003%**		90-95%	209	**0.33%**	
	>95%	63316	**99.99%**		>95%	62787	**99.16%**	

Soil Structure	<75%	0	**0**	84	<75%	219	**0.35%**	0
	75 - 89%	4	**0.01%**		75 - 89%	660	**1.04%**	
	90-95%	66	**0.10%**		90-95%	4940	**7.80%**	
	>95%	63247	**99.89%**		>95%	57499	**90.81%**	
Igneous landscape	Observational records				Detailed plots			
	% Similarity Class	km^2	**% total area**	Minimum %similarity	% Similarity Class	km^2	**% total area**	Minimum %similarity
Climate	<75%	0	**0.0%**	84	<75%	1	**0.02%**	60
	75 - 89%	29	**0.3%**		75 - 89%	178	**1.9%**	
	90-95%	196	**2.1%**		90-95%	1788	**19.5%**	
	>95%	8944	**97.5%**		>95%	7202	**78.5%**	
Vegetation density	<75%	1	**0.01%**	59	<75%	5	**0.06%**	5
	75 - 89%	2	**0.02%**		75 - 89%	128	**1.4%**	
	90-95%	11	**0.1%**		90-95%	149	**1.6%**	
	>95%	9154	**99.9%**		>95%	8886	**96.9%**	
Soil nutrient	<75%	7	**0.1%**	35	<75%	69	**0.8%**	27
	75 - 89%	26	**0.3%**		75 - 89%	129	**1.4%**	
	90-95%	142	**1.6%**		90-95%	219	**2.4%**	
	>95%	8844	**98.1%**		>95%	8602	**95.4%**	
Soil structure	<75%	0.2	**0.002%**	68	<75%	18	**0.2%**	48
	75 - 89%	51	**0.6%**		75 - 89%	422	**4.7%**	
	90-95%	284	**3.2%**		90-95%	1166	**12.9%**	
	>95%	8682	**96.3%**		>95%	7412	**82.2%**	

Beta-diversity

The survey design reliably captured the beta-diversity on both landscapes with the mean beta-diversity for each landscape within the 95% confidence intervals (Table 2.2). On both landscapes the dissimilarities were strongly skewed to the right (Figure 2.1), suggesting that the tight confidence intervals around the mean were not unexpected as any randomly sampled subset has a high probability of selecting plots with dissimilarities close to the mean. The mean dissimilarity was higher for the igneous landscape, indicating a higher degree of change in species composition between plots, hence a greater beta-diversity and heterogeneity of plant communities (Havlová et al., 2004).

Table 2.2. Beta-diversity measures with 95% confidence intervals of the Tertiary and igneous landscapes of CYP. Confidence intervals were calculated from average Bray-Curtis dissimilarities from 50 random sample subsets on each landscape.

	Mean Bray-Curtis dissimilarity	95% Confidence Interval
Tertiary landscape	77.7	77.6 - 78.0
Igneous landscape	85.3	85.1 - 85.4

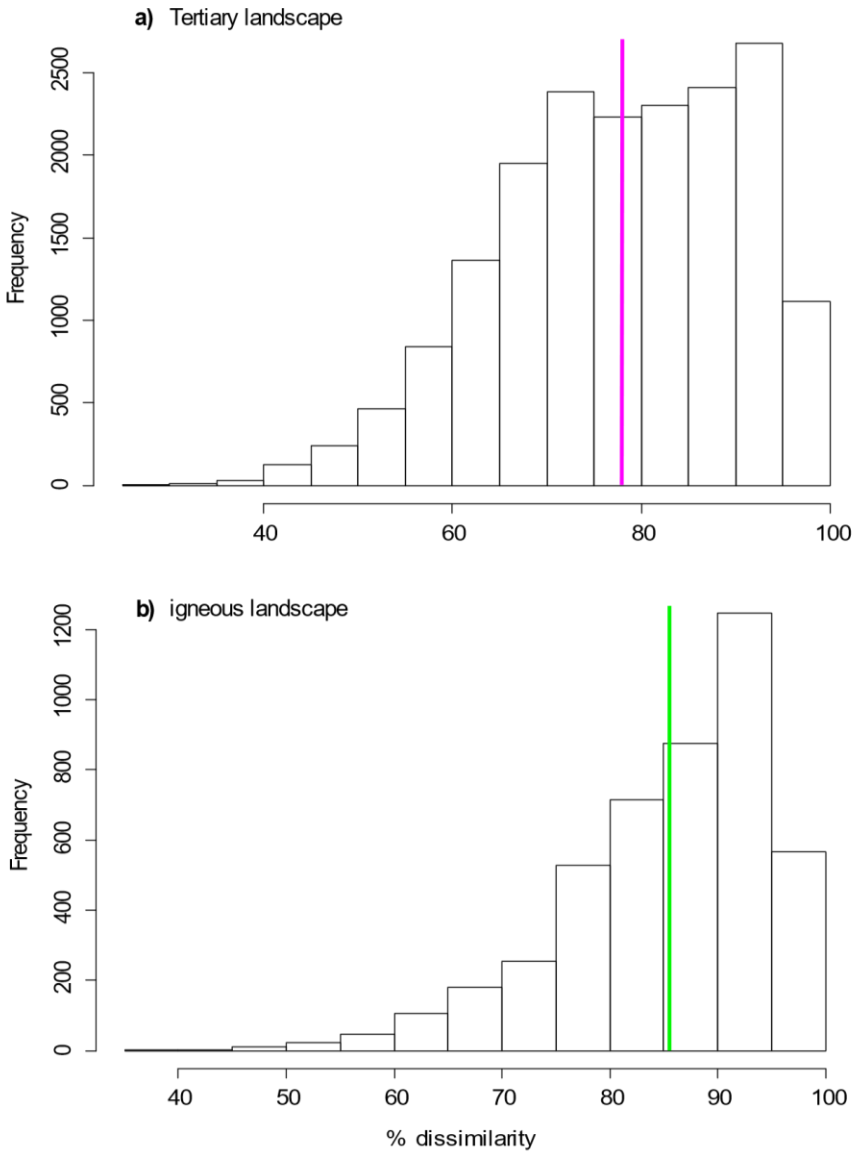

Figure 2.1. Histogram of pairwise dissimilarities between all detailed plots on a) the Tertiary landscape and b) the igneous landscape. Vertical line = mean dissimilarity, representing beta-diversity.

Species richness

The Government's survey design did not reliably capture the full species richness on either landscape. The species accumulation curves for both landscapes indicate that an asymptote has not been reached and more species will be captured with more plots (Figure 2.2). This is supported by the species richness estimators which shows the number of species captured by the detailed plots does not overlap with the variance of any of these (Figure 2.3). The

estimates of species richness are between 1.2 times and 1.6 times greater than the number captured on both landscapes.

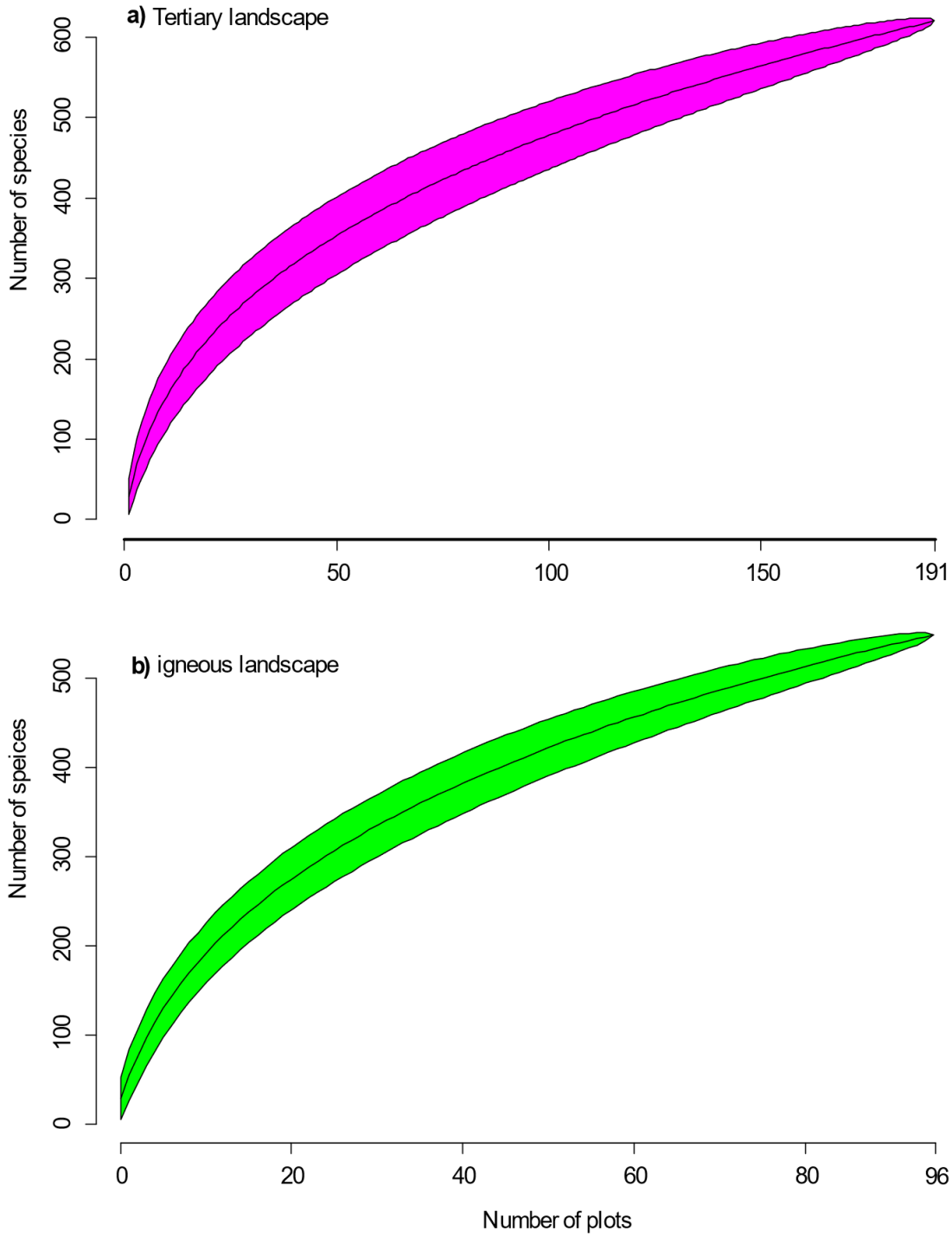

Figure 2.2. Species accumulation curves for each landscape, with standard deviation.

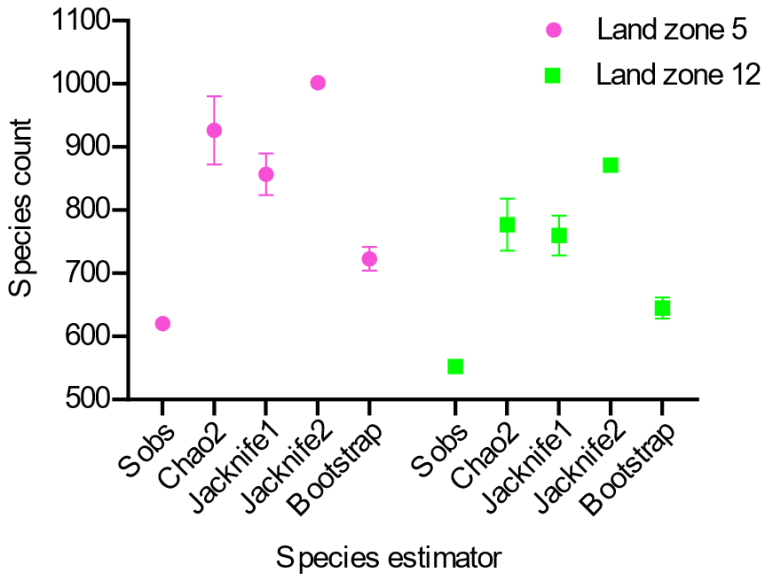

Figure 2.3 Plots of species richness estimates, with standard errors, compared with sampled species richness (Sobs) for each landscape.

Discussion

The survey design adopted by the Queensland government is preferential in that detailed plots are sited at locations which are representative of the surrounding community. The locations are determined from observational records taken during traverses through the landscape. I assessed how well this survey design captured the environmental variability, the beta-diversity and the species richness of the Tertiary and igneous landscapes of CYP, an area of 51 500 km^2. I found that with a total of 288 detailed plots the design did not capture the species richness of the either landscape adequately but did comprehensively capture both the environmental variability and beta-diversity.

Not capturing the species richness adequately agrees with the intuitive assessment that sampling such a large area with 288 detailed plots would not provide comprehensive coverage; and with Lawson, Ferrier, Wardell-Johnson, Beeton, & Pullar (2010) who found high levels of floristic heterogeneity within regional ecosystems in south-east Queensland. This is in contrast to other studies that found preferential-sampling had a higher likelihood of sampling full species richness than stratified-random sampling, as researchers tend to choose sample locations with higher species richness (Michalcová et al., 2011). In this

survey, however, locations were chosen on a perception of representativeness of distinctive communities, rather than species richness, potentially explaining the difference to other studies. Species richness is an important component of diversity and often used to determine conservation priorities and outcomes (Margules & Pressey, 2000). However, documenting the full species richness across extensive areas is dependent on comprehensive spatial and temporal coverage and requires resources beyond the capability of most organisations. It is unrealistic to expect a landscape classification system to capture the full species richness. The aim of the RE system is to use plant communities (the beta-diversity) as surrogates for biodiversity for conservation planning and it is therefore more important that the survey design adequately captures the beta-diversity of the landscape.

The sampling design captures the environmental variability well (<5% of both landscapes inadequately sampled) and within this surveyed environmental variability, the design also captures the beta-diversity comprehensively. Although it is possible that inadequate sampling of species richness may lead to inadequate capture of the beta-diversity in a landscape if species characterising plant communities are not included in the data (Magurran, 2004; Roden et al., 2018), adequate capturing of species richness is driven by sparse species (Chiarucci et al., 2003), and Roden et al. (2018) show that beta-diversity is well captured by dominant species. As the criteria for identifying plant communities within the RE system are based on dominant species, and plot locations are designed to capture these, it is unlikely that the low sampling of species richness would limit the capture of beta-diversity. Beta-diversity is, rather, linked to environmental variability, with higher variability leading to a larger number of niches for unique plant communities to inhabit (Kent, 2012). The capture of the beta-diversity in the landscape is therefore dependant on the adequate sampling of the environmental variability and preferential sampling designs cover a greater range of environmental extremes than random sampling designs for the same level of survey effort (Roleček et al., 2007). I posit that the two-tiered system of data collection in the Government's design, with observational records being used to determine that the locations

of detailed plots to ensure representativeness, provides this. Comprehensive coverage of the environmental variability is however dependant on access. The areas of environmental variability not well sampled in these landscapes correspond to areas that are either rainforest (and outside the scope of the dataset in this project) or areas not surveyed due to accessibility issues such as distance to roads or lack of ability to land helicopter (Appendix 2.1). The capture of environmental variability by detailed plots was lower than that of the observational records (between 9 – 19%) and I suggest this is because collecting observational records using techniques such as aerial survey by helicopters is possible in areas which are inaccessible for collection of detailed plots. I would expect that new plant communities would be found in areas where the environmental variability is not well sampled, and these should be targeted for future survey work.

Conclusions

Preferential-sampling designs are biased in several ways compared to stratified random-sampling designs (Diekmann et al., 2007; Hédl, 2007; Michalcová et al., 2011) and it is well recognised that the statistical power of preferential-sampling designs is lower (Lájer, 2007). However, much of the aim of vegetation survey and mapping is to distinguish and describe patterns rather than produce inferential results based on null hypothesis significance testing (De Cáceres et al., 2015).The survey design adopted by the Queensland Government comprehensively captures the environmental variability and the beta-diversity present within a landscape, but not the full floristic variability within those communities. While the RE system therefore does not well predict the distribution of all species in the landscape, it does fulfil the primary function of a broad scale classification system: identifying the major ecological patterns in a landscape.

Acknowledgments

This work was carried out with the support of the Queensland Herbarium, Department of Environment and Science, Queensland Government. I thank Peter Bannink for the map

figures and Andy Gillison for help and advice running the DOMAIN software and Will Edwards for statistical advice.

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Chapter 3 Determining appropriate class definition procedures to form a new classification approach in the RE system

Contextual overview

Having established the adequacy of the dataset underpinning the RE system in capturing environmental variability, community diversity and species richness, the next step in advancing a new classification approach is to determine appropriate quantitative-based class definition procedures. In this chapter I determine suitable primary vegetation attributes and un-supervised techniques for identifying plant communities in the RE system to incorporate into the new procedures. This chapter is based on a literature review (Appendix 3.1) and a published paper; **Addicott, E**., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7. Here I use a larger dataset than in Chapter 2 as plots in homogenous vegetation types were required, which meant excluding those in ecotones, whereas in this chapter it is preferable to include all available detailed plots in analysis.

Statement of contribution of others to the published paper:

Addicott conceived the idea, performed the analysis and wrote the paper. Laurance provided intellectual support and help writing the manuscript. Lyons provided the analysis using Generalised Linear Modelling and the AIC information metric, help interpreting the results and editorial assistance. Butler provided intellectual support and comments on the manuscript. Neldner collected much of the original data and commented on the manuscript.

Introduction[1]

A critical part of a classification system is the class-definition procedures used to identify the vegetation types within the system. Choosing appropriate techniques for a classification system involves two critical parts of the class definition procedures: the primary vegetation attributes and the plot-grouping and evaluation techniques. Primary vegetation attributes are the 'subset of plants of interest', determined by the concepts and criteria of the classification system, and, the attributes of the plants such as structure, abundance or taxonomic level (De Cáceres et al., 2015). There is now a plethora of un-supervised plot-grouping and internal evaluation techniques, along with an extensive literature, and continual development of new techniques to accompany the numerous vegetation classification systems around the world. However, if the plant communities identified using quantitative based class definition procedures are to be consistent with the concepts of the RE system, then the primary vegetation attributes and un-supervised techniques will also need to be consistent with the criteria outlined in the Queensland Methodology (Neldner, Wilson, et al., 2019), including the requirement for mappability at a landscape scale.

Maps showing the extent and distribution of plant communities across large areas of the landscape are a management tool commonly associated with vegetation classification systems (J. Franklin, 2013). Maps are used for exploration of spatial and temporal changes (Accad, Neldner, Kelley, & Li, 2017) and ecological patterns of species distribution (P. J. Clarke, Knox, Bradstock, Munoz-Robles, & Kumar, 2014; Kent, 2012) and provide a predictive role in describing the distribution of plant communities in inaccessible areas. Map development involves extrapolating from areas of a specific imagery pattern with known plant communities to areas of the same imagery pattern and unknown communities (J. Franklin, 2013). When mapping extensive landscapes, differences are distinguished by

[1] This section includes part of the published paper **Addicott, E**., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7

changes in the dominant species canopy cover, by vegetation structure and by geomorphological differences in the landscape (J. Franklin, 2013; Küchler & Zonneveld, 1988; Pedrotti, 2013).

The identification of plant communities is largely dependent on purpose and scale (Gillison, 2012). For maps to be widely applicable the identification of plant communities needs to be commensurate with the scale of changes delineated in the mapping. Plot-based inventories of species assemblages are often used as part of the mapping process to describe map units (i.e. plant communities) and may also be used to derive or test vegetation classification systems applied through mapping. For a quantitative, plot-based, classification exercise using multivariate species data to be relevant to the mapping process, it needs to incorporate the attributes used to differentiate mapped changes. Across extensive landscapes this means changes in species canopy cover and vegetation height. These may be influenced by recurrent disturbance patterns, such as past land management practices. In savanna vegetation, fire history is particularly important as it can influence species assemblages and the structure of plant communities across the landscape (Miller & Murphey, 2017). Therefore, communities need to be distinguished by species that respond to, and are indicative of, landscape scale changes rather than short-lived temporal dynamics or change driven by small scale phenomena such as micro-climatic differences.

Plot-based classification exercises using full species inventories will include non-dominant, occasional species in a dataset (here termed rare). However, the distribution of these rare species is difficult to predict for many possible reasons. For example, rarity may be because species are responding to localised variations in the environment below the scale of mapping (Kent, 2012) or to past landscape disturbance history such as fire regimes. Species may also be rare in the dataset due to biases resulting from sampling designs (for example, seasonality). Thus, they contribute to ‘noise’ in the dataset from the view point of broad-scale vegetation classification systems, possibly masking the relationships of interest between vegetation samples at landscape levels (Kent, 2012) and leading to plant communities

defined at, and characterised by species responding to habitat changes at, scales below that of the mapping. This compromises the application of both the map and the quantitative classification system as ecologists lose confidence in both if the plant communities do not relate to plausible ecological interpretation at the mapping scale. Removing rare species that contribute to 'noise' in the dataset is often recommended and decisions on rarity are commonly based on frequency of occurrence (Kent, 2012; McCune & Grace, 2002). This, however, can be problematic in broad-scale mapping projects with vegetation plot locations chosen using a preferential sampling design. Such sampling designs may result in map units, distinctive in terms of species and/or structure at the appropriate scale, being represented by single plots. As a result, species dominating communities represented by single plots may occur once or twice in the dataset, and, if rare species are chosen based on low frequency, these dominant species are removed. The consequence is losing essential information about plant communities in the mapping and risking misclassification of their representative plots. An alternative is to remove species with rarity measured as consistently low contribution to cumulative abundance (Field, Clarke, & Warwick, 1982; Grime, 1998; Mariotte, 2014).

Mapped plant communities identified using both floristic and structural components have the broadest application in both research and planning (Küchler & Zonneveld, 1988). Vegetation structure is a well-established feature for differentiating vegetation at landscape scales and is represented both vertically by vegetation layers within a community and horizontally by change in vegetation formations across the landscape (Küchler & Zonneveld, 1988). Height of vegetation layers is commonly used in classification systems to represent this; for example, in Australia vegetation is classified using vegetation formations defined partly by layer height (Executive Steering Committee for Australian Vegetation Information & Department of the Environment and Heritage, 2003; Hnatiuk, Thackway, & Walker, 2009) whilst in other countries authors may weight species by transformations of layer height (Hall, 1992; Leathwick, Wallace, & Williams, 1988).

Identifying plant communities using plot-based data was historically carried out using expert knowledge to allocate plots into groups of similar vegetation (Whittaker, 1973b). With the advent of computers statistical and mathematical models have become established techniques for grouping plots with high similarity to form clusters representing vegetation types (Goodall, 2014; Kent, 2012). There has been a multiplicity of these techniques developed (Chytrý et al., 2019; Goodall, 2014; Peet & Roberts, 2013), however the choice of which one to use can influence the vegetation types identified (De Cáceres et al., 2018; Wiser & De Cáceres, 2013). The techniques most commonly used are either hierarchical clustering or non-hierarchical partitioning (De Cáceres et al., 2018; De Cáceres & Wiser, 2012; Kent, 2012). Hierarchical clustering techniques assess the similarity of individual plots in a hierarchical manner. They generally produce a dendrogram representing a hierarchy of the dis/similarity between plots (Goodall, 1973; Kent, 2012) with cutting the dendrogram at a given level of similarity forming clusters. Non-hierarchical partitioning assesses plots sitting close to each other in multi-dimensional space (Kent, 2012) and partitions it so that plots close to each other are considered a cluster (Kent, 2012). Clusters in both techniques are taken to represent vegetation types. Agglomerative hierarchical clustering techniques are the most commonly used in vegetation classification exercises and are un-supervised techniques which allow ‘natural’ groupings to be identified (Kent, 2012). Non-hierarchical techniques can measure ‘goodness-of-fit’ of plots to clusters, allowing the ecologist to understand how similar any plot is to an existing community, thus identifying possible new communities (Wiser & De Cáceres, 2013) while acknowledging plant species are distributed along an environmental continuum (De Cáceres, Font, & Oliva, 2010; Mucina, 1997). One of the major considerations in the development of all techniques is the concept of robustness, where the results of the classification are not dependent on the underlying structure of the data (Kent, 2012).

Evaluating the identified plant communities as fit-for-purpose is an essential part of the class-definition procedures and there are two types of evaluators; internal and external

(Gauch & Whittaker, 1981). Internal evaluators rely on criteria which assess clusters (plant communities) using the primary vegetation attributes and the cluster's compositional characteristics and are most commonly used to decide the level of cluster division which form communities (Gauch & Whittaker, 1981; Peet & Roberts, 2013). Internal evaluators are either geometric or non-geometric with geometric evaluators based on comparing the similarity of plots within and between clusters and non-geometric evaluators based on the strength of species' association with clusters. Clusters chosen by one or other method may be contrastingly different (Aho, Roberts, & Weaver, 2008). External evaluators are factors outside the clustering analysis, such as environmental gradients, and are more often used to validate the final clusters as plant communities (De Cáceres et al., 2015; Gauch & Whittaker, 1981). Expert recognition of the vegetation types represented by cluster groups is, however, emphasised as the most important consideration in evaluating plant communities recognised by a classification exercise (Goodall, 1973; Kent, 2012; Lötter, Mucina, & Witkowski, 2013; Whittaker, 1973a), as end-users need to be able to recognize plant communities in the landscape.

There is no general agreement on the most suitable primary vegetation attributes or plot-grouping and internal evaluation techniques for vegetation classification exercises (De Cáceres et al., 2015; Koci, Chytrý, & Tichý, 2003; Wesche & von Wehrden, 2011) with the number of techniques available complicating their application. Those used need to be consistent with the concepts and criteria of the classification system. There is also no general agreement on preferable internal evaluators to decide cluster division levels and comparing the classification outcomes produced by both geometric and non-geometric types is recommended (Aho et al., 2008; Lötter et al., 2013; Peet et al., 2018).

In this chapter, I review and recommend appropriate primary vegetation attributes and un-supervised techniques for identifying communities within the RE classification system. Choosing appropriate primary vegetation attributes requires consideration of four factors. These are: the abundance measure to use, the vegetation layers to include, the subset of

species to use (De Cáceres et al., 2015) and how to include vegetation structure. Recommendations for the first two of these (considered in the Discussion section) come from a review of the literature and attention to the concepts and criteria of the RE classification system. Recommendations for the last two result from the publication of a quantitative assessment in which I specifically investigate two questions: how does 1) removing rare species based on contribution to total foliage cover, and 2) weighting species cover by different measures of vegetation layer height, influence the classification outcomes of plant communities in tropical savanna vegetation on Cape York Peninsula and published (Addicott, Laurance, et al., 2018). To recommend suitable un-supervised plot-grouping techniques I trialled two as a result of reviewing the literature. I compared and contrasted the plant communities identified by a hierarchical technique (choosing agglomerative hierarchical clustering as it is the most commonly used) and the non-hierarchical technique of fuzzy-noise clustering (because of its potential outputs and the work of Wiser and De Cáceres (2013) and Tichý, Chytrý, and Botta-Dukát (2014). I trialled these techniques with preliminary parameters in the expectation of future comprehensive testing, however due to Government resourcing restraints I was requested not to continue. The comparison between the two techniques is therefore illustrative, and refinement of the parameters used in the fuzzy-noise clustering technique would be required for comprehensive testing between methods to be carried out. My recommendations for internal evaluation techniques were based on a review of the literature (considered in the Discussion section). The resulting suite of recommendations was considered by the Queensland Government for adoption as Government practice at a peer-review workshop.

Methods

Determining the appropriate 'subset of species' and incorporating vegetation structure[2]

Study area

This study encompasses the savanna vegetation occurring on the igneous landscape of CYP (ranges, hills and lowlands formed from Mesozoic to Proterozoic igneous rocks) (Figure 1.2).

Data collation

Detailed plots were deleted if they contained taxa identified only to family level which contributed >1% of TFC to a layer. This left a total of 101 plots comprising three main formations: grasslands (n = 14 plots), shrublands (n = 21 plots), and woodlands (n = 66 plots). From this plot data I compiled two different datasets to test for effects on plant community identification. The first, called 'cover', I used to test for the effects of rare species based on contribution to total foliage cover (TFC). My 'cover' dataset used species only from the canopy layer. In woodlands this was the tallest tree layer, in shrublands the shrub layer and in grasslands the ground layer. This formed a dataset of 101 plots and 247 species with grasslands having 137 species, shrublands 80 species and woodlands 66 species (Table 3.1). The second dataset, called 'height', I used to test for the effect of vegetation height. This dataset was 78 plots and 265 species (Table 3.1). I used the same 78 plots used to define the pre-existing communities to allow comparisons with my final classification outcomes (work that was specific to another project and not included here). Fourteen plots were grassland, 16 were shrubland and 48 were woodland. This dataset included species in the canopy layer plus all other woody dominated layers with TFC of 10% or more (Neldner, Wilson, Thompson, & Dillewaard, 2012). Species in the 'height' dataset were excluded (from

[2] This section is based on the published paper **Addicott, E**., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7

each layer) based on my analysis of rare species contribution to TFC (grasslands <8% of TFC, shrublands <1%, woodlands <10% of TFC).

Using my 'cover' dataset I explored the effects on classification outcomes of removing rare species (defined here as their contribution to TFC) by defining four rarity thresholds; 1%, 5%, 8% and 10% contribution to TFC. These were determined a priori through an expert panel of regional mapping specialists. I created four data subsets; C>1 = species contributing >1% to TFC included, C>5 = species contributing >5%, C>8 = species contributing >8%, and C>10 = species contributing >10% to TFC included. The dataset consisting of the full species pool I termed ALL. Excluded species were below threshold levels for all plots and resulted in changes in community structure (Table 3.1). Following the advice of M. J. Anderson et al. (2011) I calculated beta diversity as variation in community structure amongst my samples using Whitaker's beta-diversity calculation.

Table 3.1 Parameters and diversity of datasets. Subsets result from removing species based on % contribution to total foliage cover: ALL = full species pool, C>1 = only species contributing >1% to total foliage cover (TFC), C>5 = species >5% to TFC, C>8 = species >8% to TFC, C>10 = species contributing >10% to TFC; NoHeight = dataset used to weight species by height of vegetation layer. α = mean number of species per plot, β_w = Whitaker's beta diversity ((Total number of species / α) – 1), MSP_m = mean Marglef's species richness index per plot; ME_p = mean Pielou's evenness index per plot. Species richness values significantly different to ALL are bolded, * p < 0.001, ** p < 0.01. ^ p=0.05.

	Species	α *(st. dev)*	β_w	MSP_m *(st. dev)*	ME_p *(st. dev)*
Grasslands					
ALL	137	18 *(+/-6.96)*	6.5	4.3 *(+/-1.9)*	0.3 *(+/-0.23)*
C>1	49	9 *(+/-3.98)*	4.4	**2*** *(+/-1.1)*	0.4 *(+/-0.27)*
C>5	26	6 *(+/-2.18)*	3.3	1.3 *(+/-0.64)*	0.4 *(+/-0.28)*
C>8	16	4 *(+/-1.82)*	2.9	**0.8*** *(+/-0.56)*	0.5 *(+/-0.32)*
C>10	15	4 *(+/-1.61)*	2.9	0.7 *(+/-0.46)*	0.5 *(+/-0.32)*
NoHeight	123	16 *(+/-8.5)*	5.7	3.8 *(+/-2.41)*	0.4 *(+/-0.24)*
Shrublands					

ALL	80	9 *(+/-4.61)*	7.9	2.4 *(+/-1.39)*	0.6 *(+/-0.25)*
C>1	60	7 *(+/-3.67)*	7.6	2 *(+/-1.15)*	0.6 *(+/-0.25)*
C>5	44	6 *(+/-2.68)*	6.3	1.7 *(+/-0.86)*	0.6 *(+/-0.26)*
C>8	35	5 *(+/-2.48)*	6.0	1.4 *(+/-0.79)*	0.6 *(+/-0.27)*
C>10	31	5 *(+/-2.1)*	5.2	1.2 *(+/-0.71)*	0.6 *(+/-0.28)*
NoHeight	104	15 *(+/- 6.5)*	5.9	3.7 *(+/-2.28)*	0.6 *(+/-0.25)*
Woodlands					
ALL	66	4 *(+/-1.81)*	17.5	0.9 *(+/-0.6)*	0.6 *(+/-0.23)*
C>1	54	3 *(+/-1.56)*	16.9	0.8 *(+/-0.53)*	0.6 *(+/-0.22)*
C>5	42	3 *(+/-1.23)*	12.1	0.8 *(+/-0.45)*	0.6 *(+/-0.22)*
C>8	36	3 *(+/-1.17)*	10.5	0.7 *(+/-0.43)*	0.6 *(+/-0.22)*
C>10	33	3 *(+/-1.14)*	9.9	**0.7**** *(+/-0.43)*	0.6 *(+/-0.22)*
NoHeight	128	13 *(+/-6.2)*	10.6	2.8 *(+/-1.48)*	0.6 *(+/-0.16)*

To explore the effects on classification outcomes of weighting species by height of vegetation layer I used my 'height' dataset and four commonly used height-measures. These were; height (Height) (Hnatiuk et al., 2009; Specht, 1981); $\log_{10\,(x+1)}$ of height (LogHeight) (Hall, 1992; Wyse, Burns, & Wright, 2014); an expert-based ranking of height given to each layer (RankHeight) (Leathwick et al., 1988); and foliage cover only with no height measure (NoHeight). Height was the average height in meters of each layer in the plot. For the RankHeights the expert panel provided the following ranks based on their perception of the ecological function of each layer in the formation: woodlands and shrublands - canopy layer = 3, emergent, sub-canopy, shrub and sub-shrub layers = 2; grasslands - ground layer = 3, emergent layer = 2. To weight species I multiplied the foliage cover of each species in a layer by the height-measure of the layer. Weighted species were summed across layers to give a total value per plot.

Weighting species by different height-measures changed the vegetation structure within and between plots (Figure 3.1) and it is these effects on classifications that I test in this study. NoHeight, LogHeight and RankHeight up-weighted the lower vegetation layers with respect to the canopy layer (Figure 3.1). A NoHeight measure caused the most extreme change. Species in the lower layers of a plot had the same weighting as those in the canopy layer and structural differences between plots of different formations are eliminated (Figure 3.1). LogHeight proportionally up-weighted the lower layers with respect to the canopy layer and reduces the structural differences between plots (Figure 3.1). RankHeight weights species in different layers inconsistently and the outcomes are dependent on the value given by the expert panel. In addition, it eliminated all structural differences between formations (Figure 3.1). Height maintains vegetation structure both within a plot and between formations (Figure 3.1).

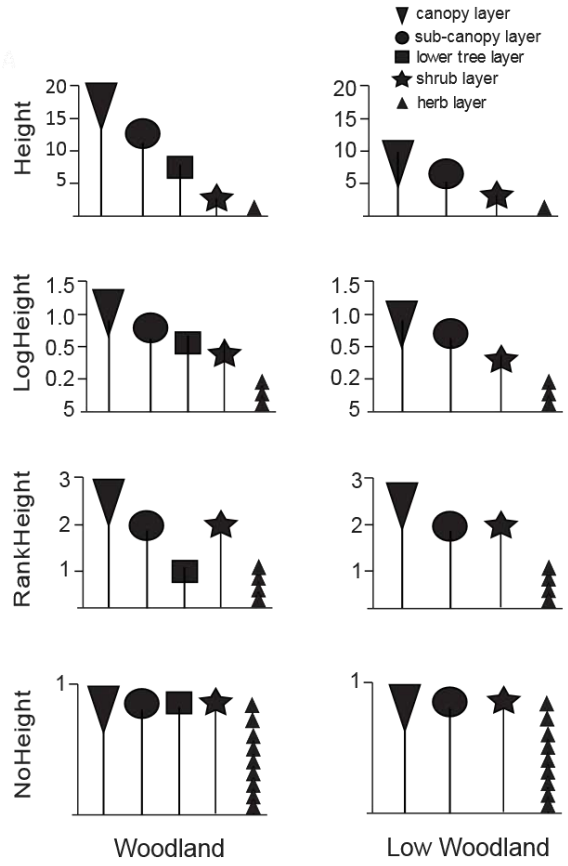

Figure 3.1 Effects on species cover of weighting by vegetation height within and between plots. The height of the symbols represents the relative weighting of each layer compared with the canopy layer. Except for Height, the height-measures up-weighted the lower layers with respect to the canopy layer within a plot and reduced or eliminated height differences between vegetation formations. I used 2 plots from the study area as my examples. Height = height in metres, LogHeight = $\log_{10(x+1)}$ of height, RankHeight = expert weightings for layers, NoHeight = no height included, foliage cover only. Vegetation layers labelled according to Ladislav Mucina, Schaminée, and Rodwell (2000).

Data analysis

I determined classification outcomes for datasets using agglomerative hierarchical clustering, and internal evaluators to determine the level of cluster division (Aho et al., 2008). All analyses were undertaken in the software package PRIMER v6 (K. R. Clarke & Gorley, 2006) or in the R environment (R Development Core Team, 2014). To test the sensitivity of my results in removing rare species I identified communities from each dataset using two common combinations of similarity measure and clustering algorithms (Appendix 3.2). These were the Bray-Curtis similarity coefficient with Unweighted Pair Means Average

linkage (UPGMA) (Kent, 2012), and Chord distance measure with flexible- β linkage (Knollová, Chytrý, Tichý, & Hájek, 2005; Nezerkova-Hejcmanova et al., 2005; Roberts, 2015). In the later I used two levels of β. Beta = -0.25 has been used effectively in numerous classification exercises (Lötter et al., 2013; Mucina & Daniel, 2013; Roberts, 2015). K. R. Clarke et al. (2014) recommend choosing a level of β that maximises the cophenetic correlation between the distance matrix and the classification dendrogram, and in my datasets β was equal to 0.01. I therefore tested changes resulting from removing rare species with three different methods: 1) Bray-Curtis similarity with UPGMA, 2) Chord distance with flexible-β at β = -0.25 and 3) Chord distance with flexible-β at β = 0.01. To determine cluster divisions, I used a combination of the SIMPROF routine ($p<0.05$) (K. R. Clarke, Somerfield, & Gorley, 2008) and Indicator Species Analysis (ISA) (Dufrêne & Legendre, 1997). The SIMPROF algorithm tests for significant difference in the between-cluster versus within-cluster similarity at each node in a cluster dendrogram, providing an objective stopping rule for cluster division (K. R. Clarke et al., 2008) in vegetation classification exercises (Oliver et al., 2012). I ran ISA in the 'labdsv' R package (Roberts, 2013). This also produced species significantly associated with a cluster ($p<0.05$) which I used as Indicator Species (IS) for each classification outcome. For the second question investigating the effects of weighting species by height-measures, I used classification outcomes resulting from the Bray-Curtis similarity coefficient and UPGMA linkage, with the SIMPROF routine to determine cluster divisions (Appendix 3.2).

I explored effects on the classification outcomes using three tests common to both questions and comparison to a baseline (Appendix 3.2). The baselines were the ALL species dataset for the first question, and the NoHeight dataset in the second question. My first test was to look for changes in the patterns of similarity or distance between plots with the 2STAGE routine in the PRIMER-e. This calculates a Spearman's rank correlation coefficient (rho) between the similarity matrices of different datasets. My second test was for differences in clustering patterns between classification outcomes. I tested for changes in proportions of

clusters per formation and plots per cluster with Fisher's exact test ($p<0.05$). One important function of a classification system is to predict patterns of floristic composition (Margules & Pressey, 2000), and so my third test, which I also used to test the quality of the classification outcomes, was to assess the ability of each classification outcome to predict the foliage cover of all species. I did this using a predictive-model based approach with generalised linear models in a multivariate framework and Akaike's Information Criterion (AIC) as an estimate of predictive performance (Lyons, Keith, Warton, Somerville, & Kingsford, 2016). In general, classification outcomes with a lower sum-of-AIC score are a better "fit" and are a way of illustrating the difference between several plausible solutions (Murtaugh, 2014). This model-based approach is available in the R package "optimus" (Lyons, 2017). When testing the removal of rare species, for each outcome from the cover thresholds I summed the AIC score across the species in the ALL dataset thus providing a measure that can be compared across classification outcomes. In both questions I used the ability to predict foliage cover to test the usefulness of the classification outcomes.

Finally, I expected removing rare species would affect community structure within my 'cover' datasets. To understand these, I tested for changes in species richness and evenness and assessed the utility of characteristic species in each formation. I calculated species richness per plot using Margalef's index (Appendix 3.2.4 Equations for indices used, and evenness of species foliage cover per plot using Pielou's index (Appendix 3.2.4 Equations for indices used. I used Margalef's index as a measure of species richness as it is independent of sample size (K. R. Clarke et al., 2014). I tested for significant differences between classification outcomes in both indices with t-tests. Characteristic species are important for identifying and describing plant communities and I tested for changes in these by evaluating the Indicator Species produced by the ISA for each classification outcome. From the IS of the ALL dataset, the expert panel nominated species responding to landscape level habitat change and therefore useful for identifying communities at mapping scales. These were termed useful-IS. For each formation in each classification outcome I tested the differences

in the proportions of total-to-useful IS using Fisher's exact test. I used this as a measure of the usefulness of the classification outcome.

Trialling plot-grouping and evaluation techniques

Data Collation

I used the detailed plots on the igneous landscape of CYP as my dataset. In line with my considerations and results on the appropriate primary vegetation attributes, I used %cover as the abundance measure and excluded ground layer species in plots dominated by woody vegetation. In woodlands I excluded, species contributing <10% to TFC at any plot, in shrublands species contributing <1% to TFC at any plot and in grasslands species contributing <8% to TFC at any plot. I incorporated vegetation structure by multiplying species cover by vegetation layer height and summing across layers. Non-native species were excluded. In this exercise all detailed plots on the igneous landscape were included with 107 plots and 177 species in total in the analysis.

Data analysis

To identify plant communities for comparison using the two plot-grouping techniques I used the software packages PRIMER-E (K. R. Clarke & Gorley, 2006), JUICE (Tichý, 2002) and the R environment (R Development Core Team, 2014).

Agglomerative hierarchical clustering

To form clusters identifying plant communities using agglomerative hierarchical clustering, I used the program PRIMER-E v6 (K. R. Clarke & Gorley, 2006) with the most common combination of transformation (square-root), similarity coefficient (Bray-Curtis) and sorting strategy (Unweighted Pair Group Mean Averaging) and grouped plots using the CLUSTER routine. I compared cluster divisions resulting from both a geometric (SIMPROF (K. R. Clarke et al., 2008)) and non-geometric evaluator (Indicator Species Analysis (Dufrêne & Legendre, 1997)). I chose the final cluster divisions using Indicator Species Analysis and the

division with highest number of significant Indicator Species (McCune & Grace, 2002). These final clusters I used as the classification outcome identifying plant communities for comparison with the classification outcome from fuzzy noise clustering.

Fuzzy noise clustering

To form clusters identifying plant communities using fuzzy noise clustering I used the R package 'vegclust' (De Cáceres, Font, Vicente, & Oliva, 2009). As the aim was to have an un-supervised classification to compare with both the agglomerative hierarchical clustering outcomes and the pre-existing supervised classification system, I used the 'incr.vegclust' routine in this package. This routine attempts to find the 'natural' number of groups by not requiring a pre-determined number of clusters as a starting point (De Cáceres et al., 2009). I calculated the distance matrix between sites using the Hellinger transformation, which calculates the square root of values which have been divided by the sum of plot values (Wiser & De Cáceres, 2018). This technique allocates plots to one of three categories, 'clearly assigned to a community', 'transitional' in which plots are transitional between communities, and 'unassigned', in which plots do not fit in to any groups and may possibly represent new communities (Wiser & De Cáceres, 2013). There are two parameters which need to be input by the user; 'm' and 'dN'. 'm' determines at what value a plot will be considered as 'transitional' between communities, with 'm' close to 1 allowing fewer plots to be transitional. 'dN' determines when a plot will be considered an outlier or a possible new community. If 'dN' is too low, then a large number of plots will be considered 'unassigned'. To identify communities at the plant association level I used 'm' = 1.2 and 'dN' = 0.75 (De Cáceres et al., 2010; Wiser & De Cáceres, 2013). I used the resulting clusters as plant communities in the classification outcome for comparison with agglomerative hierarchical clustering.

Comparing and contrasting outcomes of the two techniques

To quantify the comparison between the communities identified by each technique I compared the similarity of the central floristic concepts, and the similarity of the community compositional boundaries between them (De Cáceres et al., 2015; Tichý et al., 2014). Using the software program JUICE (Tichý, 2002) to compare the floristic concepts, I formed synoptic tables for each classification outcome, calculating a phi-coefficient of association of each species to a cluster (Chytrý, Tichý, Holt, & Botta-Dukát, 2002) (Appendix 3.3). To identify characterising species for each community I used Fisher's exact test and calculated species significantly associated with a community ($p<0.05$). Assuming that communities with highly similar characterising species described the same communities (Knollová et al., 2005), I used the 'compare two synoptic tables' routine in the JUICE software to calculate a percent-similarity of central concepts between the communities from each technique. This routine compared only the characterising species of communities with each other.

To compare the floristic composition boundaries of communities between the two techniques I used PRIMER-E (K. R. Clarke & Gorley, 2006) to form similarity matrices from community data. I firstly calculated community species data by averaging plot species data (Gauch & Whittaker, 1981), then a community similarity matrix for each classification outcome with the Bray-Curtis similarity coefficient. The important difference to measuring the floristic concepts is that these similarity matrices use all species in the dataset, rather than only the characterising species. Using the 2STAGE routine in the PRIMER-e software I calculated a Spearman rank correlation between the two community-data similarity matrices. If both the synoptic tables and the similarity matrices had high levels of agreement, then both floristic concepts and compositional boundaries would be similar between techniques (this method is described in more detail in Chapter 5 published as Eda Addicott and Laurance (2019)).

The final step in comparing the two techniques was to assess how well the communities identified by each fitted the purpose of the RE classification system. This was carried out at a peer-review workshop, so to facilitate comparisons I identified 'typical' species for each community from the agglomerative hierarchical clustering and 'characterising' species for the

communities from fuzzy noise clustering. 'Typical' species were those contributing most to the similarity of sites within a cluster and to identify these I used the SIMPER routine in the PRIMER-e software. I included species contributing >10% similarity as 'typical' (K. R. Clarke & Gorley, 2006). As 'characterising species' I used the centroid species calculated by the 'vegclust' R package (De Cáceres et al., 2009).

Peer-review workshop process

My recommendations were considered by the Queensland government at a Government sponsored peer-review workshop. Participants were ecologists with expertise in mapping regional ecosystems, multivariate ecological analysis, pre-existing communities defined using supervised techniques, the vegetation of the study area and the RE classification system.

Participants discussed each of my recommendations, assessing the outcomes from the primary vegetation attributes and the plot-grouping techniques separately. The process involved comparing and contrasting: 1) the allocation of plots to communities from the two un-supervised and the supervised techniques; and 2), the proposed communities from each un-supervised technique using external evaluation criteria to assess ecological interpretability (Appendix 3.5). These external evaluators consisted of a standard set of criteria; landform, geology, other vegetation data, soils, geographical distribution and the concepts and criteria of the RE system (Appendix 1.1).

Results

Appropriate 'subset of species' and incorporating vegetation structure[3]

Classification in the absence of rare species

Removing rare species that contributed up to 10% to TFC did not significantly change the patterns of similarity or distance between plots (Spearman's rank, ρ >= 0.95). There were

[3] This section is based on the published paper **Addicott, E.**, Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications

however slight differences between formations using Bray-Curtis similarity, with the largest apparent effect in the more species-rich grasslands. There were no differences between formations using Chord distance measure (Table 3.2). These outcomes were substantiated by my result that the species evenness of plots did not change with removal of rare species (Table 3.1).

Table 3.2 Spearman rank correlations between the Bray-Curtis coefficient and Chord distance matrices of the ALL dataset (the full species pool) and each data subset in each formation. C>1 = only species contributing >1% to total foliage cover (TFC), C>5 = species >5% to TFC, C>8 = species >8% to TFC, C>10 = species contributing >10% to TFC.

Data subset	Grasslands	Shrublands	Woodlands
Bray-Curtis similarity coefficient			
C>1	0.98	1.00	1.00
C>5	0.96	0.99	0.99
C>8	0.94	0.98	0.98
C>10	0.93	0.97	0.97
Chord distance measure			
C>1	1.00	1.00	1.00
C>5	1.00	1.00	1.00
C>8	0.99	0.99	1.00
C>10	0.99	0.98	1.00

Given these results, it was unsurprising that the difference in the number of communities was not significant, either between any data subset, or tested methods of plot-grouping techniques. Removing rare species did, however, have two important effects on classification outcomes. Firstly, it increased the detection of useful-Indicator Species and

and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7

secondly, provided as good as, or better, a prediction of foliage cover of the full species pool. In the grasslands, the species richness declined significantly, firstly between the baseline dataset (ALL) and C>1 ($t = 4.27$, $p<0.001$) and then again between C>1 and C>8 ($t = 4.34$, $p<0.001$) (Table 3.1). These declines in species richness increased the proportion of useful-IS significantly, although for different data subsets in each method (Table 3.3). The ability of the clusters from each classification outcome to predict the foliage cover of the full species pool differed between methods. With UPGMA the data subsets reduced the number of clusters identified (Appendix 3.2) but improved the ability of clusters to predict foliage cover, with C>8 subset providing the best prediction (Figure 3.2). The clusters identified with the flexible-β method were the same in each data subset and so were equally as good as ALL in predictive ability. In the shrublands the decline in species richness between ALL and each subset was not significant until C>10 ($t = 2.89$, $p<0.01$) (Table 3.1). Again, the proportion of useful-IS rose, although these proportional changes were not significant (Table 3.3). The ability of clusters to predict species foliage cover differed between methods. Again, the flexible-β method identified the same clusters in all datasets, and so all subsets predicted the foliage cover of the full species pool equally. The UPGMA method reduced the number of clusters identified from seven to six (C>1) and then to five (C>10) (Appendix 3.2) resulting in improvements in predicting foliage cover when compared with ALL (Figure 3.2). However, it was C>1 subset which had the best predictive ability (Figure 3.2). The woodlands differed from the other two formations in that removing rare species changed the patterns of clustering in the same way with all methods (Appendix 3.2). There was no consistent decrease in the number of clusters despite declines in species richness, which became marginally significant at C>10 ($t = 1.93$, $p = 0.05$) (Table 3.1). In contrast to the other two formations, all datasets had >90% useful-IS (Table 3.3). None of the datasets was better at predicting species foliage cover than any other (Figure 3.2).

Table 3.3 Number of Indicator Species (IS) and useful-Indicator Species (useful-IS) in each data subset from each method. UPGMA = Bray-Curtis coefficient and UPGMA linkage; β = -0.25 = flexible-β linkage and Chord distance measure with β = -0.25, β = 0.01 = flexible-β linkage and Chord distance measure with β = 0.01 - chosen to maximise the cophenetic

correlation between the dendrogram and the distance matrix. ALL = full species pool, C>1 = only species contributing >1% to total foliage cover (TFC), C>5 = species >5% to TFC, C>8 = species >8% to TFC, C>10 = species contributing >10% to TFC; Significant differences between ALL and subsets in bold, *p < 0.01, **p = 0.02

	UPGMA		*β* = -0.25		*β* = 0.01	
	IS	useful-IS	IS	useful-IS	IS	useful-IS
Grasslands						
ALL	45	15	23	11	16	8
C>1	25	14	11	9	8	**8****
C>5	15	10	7	**7****	6	6
C>8	11	**9***	7	7	6	6
C>10	10	9	7	7	6	6
Shrublands						
ALL	80	14	24	18	14	13
C>1	60	13	19	17	14	13
C>5	44	11	16	15	13	12
C>8	35	14	15	14	13	12
C>10	31	11	13	12	11	10
Woodlands						
ALL	10	10	14	13	17	16
C>1	10	10	15	14	16	16
C>5	10	10	15	14	15	15
C>8	10	10	15	14	16	16
C>10	12	12	15	14	15	15

Inspection of the original data revealed two reasons for the changes in proportions of useful-IS between datasets. The first was that members of the expert panel had nominated species if they were useful for identifying communities across all landscapes in Cape York Peninsula,

not just those on the igneous landscape of my study. Consequently, any Indicator Species useful for other landscapes were eliminated by the analysis, due to rarity in my dataset. Secondly, consequent to the removal of rare species those nominated by the expert panel as useful moved from being non-Indicator to Indicator Species in the analysis.

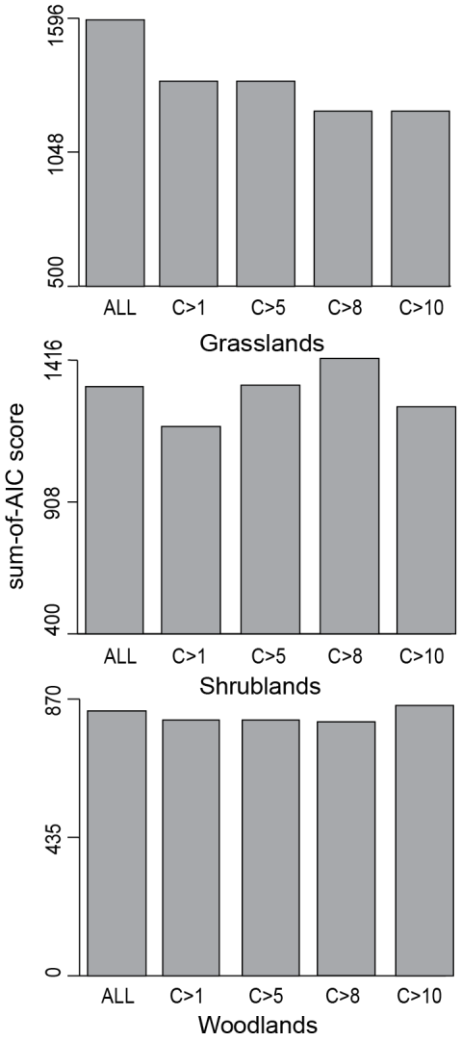

Figure 3.2 Predictive ability of classifications resulting from removing species based on % contribution to total foliage cover (TFC). Species subsets were formed by removing species whose contribution to TFC was below a threshold %. The resulting classification from each subset was used to test how well it predicted the foliage cover of all species using a zero-inflated beta regression model (Lyons et al. 2016). The lower the sum-of-AIC score the better the predicative ability. Species subsets: ALL = full species pool, C>1 = only species contributing >1% to TFC, C>5 = species >5% to TFC, C>8 = species >8% to TFC, C>10 = species contributing >10% to TFC. Only results from clustering with Bray-Curtis and UPGMA clustering are shown as there was no difference between datasets using flexible-β clustering.

Classification with species weighted by vegetation layer

Weighting species by the four different height-measures changed the patterns of similarity between plots (Table 3.4). NoHeight was least correlated with Height reflecting the maintenance of full vegetation structure using Height and the complete elimination of structure using NoHeight (Figure 3.1). NoHeight was most strongly correlated with RankHeight reflecting that both treatments minimise height differences between formations.

Table 3.4 Spearman rank correlation between similarity matrices of each height dataset. Similarity matrices were calculated using the Bray-Curtis coefficient. Height = height in meters, LogHeight = log10 (x + 1) of height, RankHeight = expert weightings for layer.

	NoHeight	RankHeight	Height
RankHeight	0.99		
Height	0.87	0.88	
LogHeight	0.91	0.88	0.95

Including height changed how different vegetation layers drove clustering in each classification outcome and substantially improved the prediction of species foliage cover (Figure 3.3). The influence of layer in plot clustering resulted in different communities defined by the clusters (Appendix 3.2). The size of these changes differed between formations with the largest in the woodlands, whereas in the grasslands and shrublands it changed the number of clusters only slightly, if at all (Table 3.5). In the woodlands, Height grouped plots emphasising firstly the canopy then the sub-canopy layer. NoHeight, in contrast, clustered plots with more emphasis on the sub-canopy and shrub layers while LogHeight and RankHeight both clustered plots with inconsistent emphasis on different layers. The plots which changed clusters between height-measures were those with high cover in multiple layers, reflecting the up-weighting of species in the lower vegetation layers by all measures except Height (Figure 3.1). In the shrublands and grasslands LogHeight clustered plots by emphasising the emergent layer, while all other height-measures clustered plots

emphasising the canopy layer. Importantly, Height best predicted foliage cover, while NoHeight was worst. LogHeight was better at predicting foliage than RankHeight (Figure 3.3).

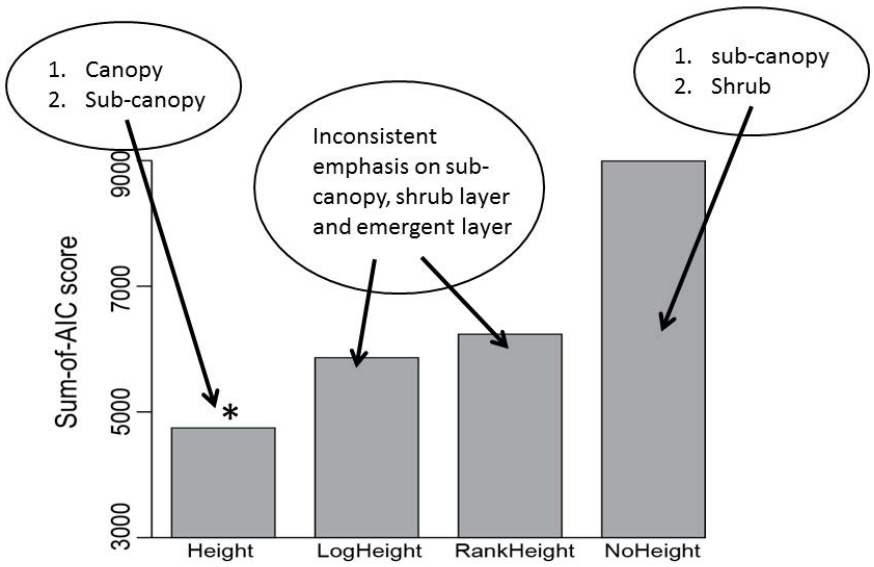

Figure 3.3 Predictive ability of classifications and the vegetation layers influencing clustering from each height measure. The ability of classifications from each height measure to predict all species cover was demonstrated using a zero-inflated beta regression model (Lyons et al. 2016). The lower the sum-of-AIC score the better the predictive ability. * Height is substantially better and NoHeight is substantially worse than all others. Circles indicate the vegetation layers influencing the clustering. Height emphasised the canopy and sub-canopy and NoHeight emphasised the sub-canopy and shrub layers. Height = height of vegetation layer in meters, LogHeight = $\log_{10\,(x+1)}$ of height, RankHeight = expert weightings for layers, NoHeight = no height included, foliage cover only.

Table 3.5 Change in number of clusters after weighting species by vegetation layer height. Height = height in meters, LogHeight = log10 (x + 1) of height, RankHeight = expert weightings for layers, NoHeight = no height included, foliage cover only.

Treatment	Total number of clusters	Grasslands	Shrublands	Woodlands
NoHeight	24	6	5	13
RankHeight	18	6	5	7
LogHeight	18	4	6	8

Height	15	4	4	7

Plot-grouping techniques

Each technique identified a different number of communities. Agglomerative hierarchical clustering identified 23 communities while fuzzy noise clustering identified 38 (Table 3.6, Appendix 3.4). The central floristic concepts were similar (84.8% similarity between synoptic tables) and twenty-two of the 23 groups recognised by agglomerative hierarchical clustering were recognised by fuzzy noise clustering. The one group recognised by agglomerative hierarchical clustering, but not fuzzy noise clustering, was a grassland identified by two plots. Here, fuzzy noise clustering split the sites between two different structural formations, a grassland and a woodland, neither of which community was identified by agglomerative hierarchical clustering (Table 3.6). Fuzzy noise clustering recognised 18 communities not recognised by agglomerative hierarchical clustering, many represented by singleton clusters (Table 3.6). The difference in the number of singleton site communities resulting from each technique was significant (5 versus 20) *(Fisher's exact test; p = 0.03)* (Table 3.6). The differences in the numbers of communities was reflected in the low correlation of the floristic boundaries between communities recognised by each technique (rho = 0.12).

Table 3.6 Comparison of communities from each technique. Highlighted community in agglomerative hierarchical clustering is split into two by fuzzy noise clustering (highlighted). The number of communities identified by singleton sites is significantly different between techniques. *$p=0.03$.

Agglomerative hierarchical clustering group	Typical species	No. sites	Fuzzy noise clustering group	Centroid species	No. sites
j	*Acacia polystachya* *Dodonaea polyandra*	1	F12	*Acacia polystachya* *Dodonaea polyandra* *Parinari nonda*	1
u	*Asteromyrtus lysicephala* *Neofabricia myrtifolia* *Jacksonia thesioides*	4	F18	*Asteromyrtus lysicephala* *Choriceras tricorne* *Jacksonia thesioides*	1

	Melaleuca viridiflora			*Melaleuca viridiflora*	
v	*Cochlospermum gillivraei* *Terminalia arenicola*	1	F8	*Cochlospermum gillivraei* *Canarium australianum*	1
b	*Cochlospermum gillivraei*	2	F19	*Cochlospermum gillivraei* *Eucalyptus crebra*	1
n	*Corymbia disjuncta* *Erythrophleum chlorostachys* *Corymbia clarksoniana* *Eucalyptus crebra*	3	F25	*Corymbia disjuncta*	3
m	*Corymbia nesophila* *Eucalyptus tetrodonta*	14	F36	*Corymbia nesophila*	8
i	*Corymbia stockeri* subsp. *peninsularis*	4	F33	*Corymbia stockeri* subsp. *peninsularis*	4
k	*Corymbia tessellaris*	2	F23	*Corymbia tessellaris*	1
q q	*Eucalyptus brassiana* *Corymbia clarksoniana*	3	F24	*Eucalyptus brassiana* *Corymbia clarksoniana*	3
o o	*Eucalyptus chlorophylla* *Corymbia dallachiana*	2	F20	*Eucalyptus chlorophylla*	2
r r	*Eucalyptus cullenii* *Corymbia clarksoniana*	14	F37	*Eucalyptus cullenii*	11
p p	*Eucalyptus leptophleba* *Corymbia clarksoniana*	7	F35	*Eucalyptus leptophleba*	7
l l	*Eucalyptus tetrodonta* *Corymbia clarksoniana*	13	F38	*Eucalyptus tetrodonta*	11
e e	*Heteropogon triticeus* *Sarga plumosum*	5	F32	*Heteropogon triticeus*	5
c c	*Imperata cylindrica,* *Mnesithea rottboellioides*	1	F15	*Imperata cylindrica*	1
t	*Leptospermum purpurascens*	5	F31	*Leptospermum purpurascens*	4
a a	*Lophostemon suaveolens* *Eucalyptus crebra*	1	F16	*Lophostemon suaveolens*	1
s	*Melaleuca citrolens*	3	F26	*Melaleuca citrolens*	3

w	*Melaleuca viridiflora*	6	F34	*Melaleuca viridiflora*	6
d	*Rock pavement*	2	F29	*Rock pavement*	2
f	*Schizachyrium fragile* *Aristida*	1	F17	*Schizachyrium fragile* *Aristida*	1
h	*Welchiodendron longivalve* *Acacia brassii* *Cochlospermum gillivraei*	8	F30	*Welchiodendron longivalve*	6
g	*Schizachyrium* *Atalaya hemiglauca* *Rhynchosia minima*	2			
			F21	*Acacia brassii*	2
			F14	*Acacia humifusa* *Petalostigma pubescens*	1
			F10	*Acacia leptostachya* *Eugenia reinwardtiana* *Terminalia arenicola* *Dodonaea viscosa*	1
			F6	*Allocasuarina littoralis*	1
			F27	*Asteromyrtus brassii* *Neofabricia myrtifolia* *Allocasuarina littoralis*	4
			F7	*Corymbia dallachiana*	1
			F9	*Corymbia hylandii*	1
			M1	*Corymbia stockeri* subsp. *peninsularis* *Eucalyptus tetrodonta*	1
			F22	*Corymbia stockeri* subsp. *stockeri*	2
			F13	*Ectrosia* *Eriachne*	1
			F28	*Erythrophleum chlorostachys* *Eucalyptus tetrodonta*	4
			F4	*Eucalyptus platyphylla* *Eucalyptus leptophleba*	1
			F2	*Eucalyptus cullenii*	1

				Acacia brassii	
			F11	*Melaleuca stenostachya Petalostigma banksii*	1
			F3	*Melaleuca viridiflora Choriceras tricorne Lophostemon suaveolens*	1
			F5	*Neofabricia myrtifolia Welchiodendron longivalve Melaleuca viridiflora*	1
	No. singleton communities	**5***			**20***
	No. of communities	23			38

Discussion[4]

In the quantitative assessment exercises of this chapter I have established two components of the primary vegetation attributes necessary for identifying plant communities consistent with the concepts and criteria of the RE system: the subset of species and incorporating vegetation structure. I have also shown that both agglomerative hierarchical clustering and fuzzy noise clustering are suitable plot-grouping techniques for clustering plots to identify communities. In this section I discuss these findings and consider the final components of the primary vegetation attributes (the abundance measure and the vegetation layers for inclusion in analysis) and internal evaluation techniques.

Primary vegetation attributes

Species abundance can be measured in a number of ways for classifying vegetation communities (Smartt, Meacock, & Lambert, 1974, 1976) and the Queensland Methodology focuses on three; basal area, stem counts and %cover (Neldner, Wilson, et al., 2019). While

[4] This section includes part of the published paper **Addicott, E.**, Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7

all three of these abundance measures can be used to estimate relative dominance and abundance (for example (Cavada et al., 2017; Eldridge, Delgado-Baquerizo, Travers, Val, & Oliver, 2018; Lehmann et al., 2014; Memiaghe, Lutz, Korte, Alonso, & Kenfack, 2016), adopting the appropriate measure is important in ensuring the identified communities are suitable for the classification system (Smartt et al., 1974, 1976). The arguments supporting the use of %cover as the abundance measure in the RE system are compelling. Firstly, %cover is the most globally common abundance measure used in vegetation classification systems (De Cáceres et al., 2018) and within Australia %cover is used to determine vegetation formations describing national vegetation types (Executive Steering Committee for Australian Vegetation Information & Department of the Environment and Heritage, 2003; Hnatiuk et al., 2009). Secondly, using %cover conforms with research showing it recognised geological and slope gradients better than classification exercises which used frequency or biomass, and also tended to identify the same groups as a pre-existing supervised classification system (Smartt et al., 1974, 1976). This is important as both geology and landform are structural and procedural elements of the RE system. Thirdly, the criteria in the RE system used to identify communities is %cover. An additional consideration is that at a plot level, woody vegetation species are measured in three ways while the ground layer species are measured as %cover only. Finally, %cover is the abundance measure used by existing end-uses of the RE system. For example, legislation that currently uses the RE system has cover as a determinant of legal outcomes (Queensland Government, 1999) and the RE mapping program also distinguishes communities based on cover.

The %cover contribution to TFC which resulted in classification outcomes relevant to landscape and broad-scale map communities differed between the three vegetation formations in my study and slightly between methods. However, generalised results are consistent across methods. Although grasslands are the more species rich formation in terms of the canopy layer, to identify these communities at a landscape scale species contributing <8% to TFC can be excluded and classification outcomes based on these

species can also best predict foliage cover of the full species pool. These results suggest the large majority of species in tropical savanna grasslands are responding to habitat changes at scales below those used in landscape mapping. In the woodlands, while using species at any cover level identified communities at landscape-scales, only those contributing >10% to TFC are required to both identify communities and predict species cover. The shrublands had a lower threshold; removing species contributing <1% to TFC produced useful landscape-scale classification outcomes and consistently predicted foliage cover. My results link two separate bodies of work. One demonstrates the usefulness of subsets of species data; they can improve classification outcomes in detecting major gradients (Lengyel, Csiky, & Botta-Dukát, 2012), maintain the statistical power of a dataset (Vellend, Lilley, & Starzomski, 2008), and subsets resulting from removing unidentified species continue to identify major ecological patterns from datasets (Pos et al., 2014). The other body of work shows subsets of the structural components of a community detect major ecological patterns. Mucina and Daniel (2013) found woody vegetation and dominant grasses useful in identifying savanna plant communities in north-western Australia while Nezerkova-Hejcmanova et al. (2005) found those same structural components of plant communities informative in identifying savanna vegetation types in Senegal. My findings link these in suggesting that species subsets, within structural components, can identify landscape scale ecological patterns and, I suggest, useful subsets for savanna vegetation.

As well as demonstrating techniques useful in aligning plot-based classification outcomes to broad-scale vegetation maps my work can suggest the necessary levels of sampling intensities. For instance, in the grasslands, landscape-scale classification outcomes and prediction of species foliage cover is achieved with a subset of only 34% of the total species pool, in shrublands 75% and in woodlands 50%. Understanding the level of sampling intensity required at the landscape level can indicate to ecologists which species are ‘noise’ in the dataset. Ecologists are generally counselled to take care when deciding which species to discard as they may possibly delete important characteristic species for the environmental

gradients under consideration (McCune & Grace, 2002). However, my results give confidence in understanding at what level of contribution to TFC a species may be considered noise and may also indicate when seasonally dependent annual species, often removed because they are ephemeral, might need to be included. Deleting noisy species from the dataset allows us to define a 'subset of plants of interest', an important primary vegetation attribute in vegetation classification exercises (De Cáceres et al., 2015).

Incorporating vegetation structure in identifying plant communities is useful at landscape and regional scales (Beard, 1973) and has a long history in Australia (Beard, 1973; Executive Steering Committee for Australian Vegetation Information & Department of the Environment and Heritage, 2003; Specht, 1981). Plot data collected in accordance with the Queensland Methodology includes %cover in all woody vegetation layers and the ground layer (Neldner, Wilson, et al., 2019). The considerations about including these layers in the primary vegetation attributes centre on mapping and scale issues. REs are mapped from remotely sensed imagery using techniques dependent on recognising changes in patterns of cover and dominant species, which in forested areas is the woody vegetation layers (Neldner, Wilson, et al., 2019). This method is supported by research which shows communities identified by woody vegetation are more stable for mapping (Hüttich et al., 2011), more recognisable at a landscape scale and by experts (Mucina & Tichý, 2018; Neldner & Howitt, 1991), and as informative about the distribution of all species across a landscape as communities identified using all vegetation layers (Bedward, Keith, & Pressey, 1992). At landscape scales, another major consideration is that the distribution of species in the ground layer has a low correlation with the distribution of woody species across a variety of biomes (Lewis, 2012; Neldner & Howitt, 1991; Neldner, Kirkwood, & Collyer, 2004; Nezerkova-Hejcmanova et al., 2005). This is due to different responses in dominance and species composition in the ground layer resulting from disturbance, seasonal changes and micro-climate at scales below that of landscape scale classification (Mucina & Daniel, 2013; Mucina & Tichý, 2018).

Variety of life-forms and species heights are important functional characteristics of an ecosystem (De Cáceres, Legendre, He, & Faith, 2013; Lindenmayer & Franklin, 2002; Sattler & Williams, 1999) as well as being key components in differentiating landscape scale plant communities (Küchler & Zonneveld, 1988). The RE system follows this by specifying criteria which weights the vegetation layers in a plot (Appendix 1.1) For identifying landscape-scale communities, I found using actual height of the vegetation layer was necessary as it grouped sites by canopy and sub-canopy layers and was substantially better than any of the other measures in predicting species foliage cover. This is important, as a major function of maps is in predicting plant communities across the landscape (Küchler & Zonneveld, 1988) and a plot-based classification system that best predicts species cover is likely to increase the predictive power of the mapping. My results differ from those found by Mucina and Tichý (2018) who found not including layer height was more informative for identifying plant communities in subtropical forests. My results do, however, substantiate their warning that their results may not be applicable in communities with low similarity of species between the canopy and understorey layers as is the case in savanna vegetation in north-eastern Australia.

There are necessarily subjective choices inherent in any classification exercise (Kent, 2012) and these will influence outcomes (Aho et al., 2008; Lengyel & Podani, 2015; Lötter et al., 2013; Tichý, Chytrý, Hajek, Talbot, & Botta-Dukát, 2010). To find species which indicate landscape level changes I have used species nominated by experts. Inherent in my results, therefore, is the assumption that the experts' choice of useful indicator species is also reflected in the mapping to differentiate communities.

This work allows a 'subset of plants of interest' to be defined. From this it is possible to produce a list of regionally important species for classification exercises at landscape mapping levels. This is useful for field application in directing survey time and effort at a targeted list (Marignani, Del Vico, & Maccherini, 2008). I would suggest that a plot dominated

by species not included in the ‘subset of plants of interest’ is indicative of a community new to the classification system.

Confidence of the end-users in the identification of the plant communities represented in broad-scale maps is important. A standard approach to ensuring this outcome is to test mapped communities against quantitatively identified groupings of floristic plot data. However, issues with scale, rare species and necessary attention to canopy composition and vegetation height in mapping can cause confusion between mapped communities and quantitative groupings of plot-based data. My work demonstrates that incorporating species height and removing rare species ensures that quantitatively identified communities are conceptually consistent with approaches used to identify and describe landscape patterns. This provides a tighter linkage between plot-based classification systems and remotely sensed maps, allowing more robust mapping validations (Roff, Lyons, Jones, & Thonell, 2016) and greater confidence of land managers in both the classification systems and maps.

Plot grouping and internal evaluation techniques

Both of the plot-grouping techniques trialled recognised similar floristic concepts as communities but with different compositional boundaries between those communities. Given the similarity in floristic concepts I concluded both plot-grouping techniques would be suitable in the class definition procedures in the RE system. While potential outputs from fuzzy noise clustering have advantages over those from agglomerative hierarchical clustering (De Cáceres et al., 2010), I also recognised the difference in compositional boundaries indicated an investment of time was required to gain the expertise to apply fuzzy noise clustering robustly.

There are a multiplicity of internal evaluators available to help choose the most appropriate level of cluster division for recognising plant communities (for example Aho et al. (2008) Roberts (2015) Tichý et al. (2010) M. B. Lyons et al. (2016)). The geometric evaluator SIMPROF (K. R. Clarke et al., 2008) uses permutation tests to give a statistical measure of

the similarity of plots within versus between clusters. This routine is well tested in the marine science literature using cover as the abundance measure (K. R. Clarke et al., 2014) and increasingly used in vegetation science (Hunter & Lechner, 2018; Oliver et al., 2012; Stromberg & Merritt, 2016). It is provided in the PRIMER-e software package (K. R. Clarke & Gorley, 2015) and so is easily accessible for Government practitioners. Of the non-geometric evaluators available, Indicator Species Analysis (Dufrêne & Legendre, 1997) best suites agglomerative hierarchical clustering (Aho et al., 2008) and gives information on the distribution of species across the clusters as well as being useful in choosing the final cluster division. The final cluster division is that with the highest number of significant indicator species (McCune & Grace, 2002). Although the modelling approach developed by M. B. Lyons et al. (2016) is not an internal evaluator in that it does not use the primary vegetation attributes and the cluster compositional attributes to measure cluster effectiveness, it does provide a different measure by quantifying how well different classification outcomes predict the distribution of species occurrence and abundance across the dataset. Because the different types of evaluators use quite different measures, using more than one type to provide information about communities identified at different clusters division levels is important (Aho et al., 2008). Therefore the cluster divisions resulting from these evaluators do not form the final answer but are used in the body of work considered during the peer-review process.

Class definition procedures: Recommendations and workshop outcomes

Based on my work in this chapter I presented recommendations for un-supervised class definition procedures suitable for the RE system to the Queensland Government. These were reviewed at the two-day workshop. While most of my recommendations were accepted, two were not (Table 3.7). The first one was with respect to the thresholds for excluding rare species from the classification exercise. Rather than use different thresholds of % contribution to Total Foliage Cover in different vegetation formations to form species subsets, the final outcome was to use the 1% threshold in all formations. The sentiment at

the workshop was an unwillingness to lose perceived information from the data, and my results showed that the 1% threshold was adequate (if not optimal) for all vegetation formations tested. The second outcome which differed from my primary recommendation was the plot-grouping technique. I recommended fuzzy noise clustering; however, the workshop participants chose the second option of agglomerative hierarchical clustering. Although there was general recognition that fuzzy noise clustering identified ecologically interpretable communities, considerations against its adoption were; 1) the technique is not in widespread use, especially by government institutions; 2) there was a sizable input of resources required to refine its use if it was to be adopted by the Queensland government; and 3) it was beyond the current institutional capability. The class definition procedures resulting from the workshop (Table 3.7) have been adopted throughout the rest of this thesis.

Table 3.7 Recommendations for Government practice and the outcomes from peer-review workshop

Class Definition Procedure	**Procedures recommended from this study**	**Outcome of peer-review workshop**
Primary Vegetation Attributes		
Abundance measure	%cover	Accepted
Vegetation layers to include	Exclude ground layer from woody vegetation dominated plots	Accepted

Subset of species	Grasslands: exclude species contributing <8% to Total Foliage Cover Shrublands: exclude species contributing <1% to Total Foliage Cover Woodlands: exclude species contributing <10% to Total Foliage Cover	In all formations exclude species contributing <1% to Total Foliage Cover
Species weighting to incorporate structure	Multiply each species in a plot by vegetation layer height and sum across plot	Accepted
Plot-grouping technique		
Clustering algorithm	Option 1: Fuzzy noise clustering Option 2: Agglomerative hierarchical clustering (UPGMA linkage), with square-root transformation and Bray-Curtis coefficient	Option 2 accepted
Internal Evaluators	Geometric: SIMPROF	Accepted

(to choose levels of cluster division forming communities)	Non-geometric: Indicator Species Analysis Modelling: Optimus (Linear Regressoion using AIC)

Acknowledgements

The work in the published paper **Addicott, E**., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7 was carried out with the support of the Queensland Herbarium, Department of Science, Information Technology and Innovation, Queensland Government, Australia. I thank Phil Craven, Darren Crayn, Les Mitchell and Mark Newton for discussions and comments on earlier drafts, Peter Bannink for Figure 1.2 and the 14 members of the expert panel for providing thresholds of rarity and rank heights to test.

Chapter 4 A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion

This chapter is based on a published paper (E. Addicott, Newton, et al., 2018), with content edits in the Introduction, Methods and Discussion to reduce repetition and ensure terminology is consistent with the rest of this thesis:

Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia*, 18, 29–71. doi:10.7751/cunninghamia.2018.18.003

The published paper includes photographs of representative communities resulting from my work. I have included these as an addition to Appendix 4.2 as I do not directly refer to them in the text of my thesis.

Statement of contribution of others:

Addicott conceived the idea, performed all analysis, chaired the peer-review workshop and wrote the paper. Newton provided intellectual assistance, helped with data collation and cleaning, assisted with running the workshop, and assisted with the production of tables and appendices. Laurance assisted with writing and editing. Neldner collected much of the data, provided the historical context of the work and edited the final manuscript. Laidlaw tested the methods on other datasets within government and edited the final manuscript. Butler provided intellectual input to the workshop process and direction and edited the final manuscript.

Contextual overview

The un-supervised class definition procedures put forward in Chapter 3 were determined as appropriate for inclusion in a new classification approach for the RE system. In this chapter I test them by identifying the savanna communities associated with two landscapes in the bioregion covering ~51 000 km^2, the Tertiary and the igneous landscapes. Using the new procedures I identified communities at the plant association level and developed a descriptive-framework for each community using statistically based characterising species and biophysical attributes. As a result of queries which were raised through a peer-review assessment process, I also recommended some standard quantitative external evaluation techniques. The new procedures recognised a total of 57 communities compared with 110 that had been identified using supervised techniques. These final plant communities refined Regional Ecosystems under the Queensland government's regulations and the statistically based descriptive-framework supported consistent descriptions of communities and assignment of new plots to communities. The results presented in this chapter demonstrate that the new class definition procedures can be applied at bioregional scales and are appropriate for the RE system. These procedures along with the standard quantitative external evaluation techniques and peer-review process described in this paper form a new classification approach for the RE system.

Introduction

On Cape York Peninsula (CYP) a vegetation map and supervised classification system at the plant association level was developed as part of the Cape York Peninsula Land Use Study (CYPLUS) carried out in the early 1990s (Neldner & Clarkson, 1995). With the adoption of the RE system, this vegetation classification system was converted to the RE classification system using supervised techniques. The vegetation map was also revised in the context of the state-wide RE mapping program at a scale of 1:100,000, an exercise which ultimately necessitated a revision of the RE classification system on CYP.

A classification system has widest applicability if it can perform two major tasks: firstly, determine communities with transparent and repeatable techniques, and secondly provide consistent and reliable assignment of new plots to the classification system (De Cáceres & Wiser, 2012). The aim of this study is to address these requirements by testing the new class definition procedures determined in Chapter 3 to identify appropriate plant communities within the RE system. Specifically, I aim to identify the savanna plant communities of two landscapes on CYP at the association level and develop a descriptive-framework which incorporates statistically derived characterising species. The latter will aid in assigning new site data into these communities. I use this framework to describe REs suitable as distinct vegetation mapping units.

Methods

Study area

In this chapter my study area encompasses the savanna communities on both the Tertiary and the igneous landscapes of CYP (Figure 1.2).

Dataset

Detailed plots were excluded from analysis in this chapter if they were ecotonal or located in the adjacent bioregion. This left 192 detailed plots on the Tertiary landscape and 96 on the igneous landscape. After excluding species in line with the outcomes of chapter 3 there were a total of 351 species with 241 occurring on the Tertiary landscape, 258 on the igneous landscape and 148 shared between the two.

Identifying plant communities

I analysed the detailed plots in each landscape to look for groups of co-occurring plant species using the class definition procedures outlined in Chapter 3. In situations where the internal evaluators produced differing results, I formed a subset of plots and tested cluster divisions within the subset. All analysis was done using the PRIMER-E v6 software program (K. R. Clarke & Gorley, 2006) or the R environment (R Development Core Team, 2014).

Assigning plant communities into the regional ecosystem framework

The final plant communities were evaluated by the technical review committee for regional ecosystems of the CYP bioregion. The role of this committee was to evaluate and give effect to proposals to modify RE classifications. During this process the committee assigned plant communities to regional ecosystems based on expert-judgement of non-floristic variables as outlined by the existing Queensland Methodology (Neldner, Wilson, et al., 2019), potentially producing REs containing communities with different dominant species and low floristic similarity to each other. For example, communities which did not have predictable or mappable occurrences or were <100 ha in total area of distribution were grouped with those on closely associated landforms and similar ecological niches. Communities recognised as successional temporal variants, or condition states, of a climax association were also grouped into one RE. Where the committee requested more evidence to support proposed changes, I used the criteria outlined in the Queensland Methodology as a guide for conducting further analyses. Consequently, I tested for floristic differences between plots on different geomorphological areas and soil types (using the ANOSIM routine (K. R. Clarke et al., 2008)), for differences in canopy height (using an unpaired t-test) and investigated whether differences in the ground layers of plots were coincident with geomorphological areas or soil divisions (using nMDS ordination and GIS overlay). One additional role of the committee was to identify communities not represented in the analyses but recognisable from aerial photo interpretation, non-detailed plot data and observational plots. There were therefore two types of communities in the final classification system; those identified through quantitative analysis and those identified by supervised techniques. The latter communities will be reviewed when further detailed sampling data are available.

Creating community descriptions and assigning new plots

An important aspect of a vegetation classification system is to allow description and identification of its plant communities (De Cáceres & Wiser, 2012). To this end I compiled a descriptive-framework based on characterising species, vegetation structure and landform,

including geographical distribution when it aided identification (Appendix 4.1). Characterising species were those used to describe the floristic and structural composition of a community (De Cáceres et al., 2015) and were identified for the quantitatively defined communities using each species' frequency, average cover and strength of association with a community. To determine the strength of each species' association with a community, I calculated a phi-coefficient of association (Chytrý et al., 2002) based on cover, using the JUICE software package (Tichý, 2002). Each group was standardised to equal number of plots. A phi-coefficient of 100 means a species occurs only in that community, while values approaching zero indicate the species occurs in several communities. The phi-coefficient values were also used to identify species with a significant association to a community using Fisher's exact test ($p<0.05$) (Chytrý & Tichý, 2003). I listed species frequency and average cover using the technical-description routine within the CORVEG database, which also allowed identification of vegetation structure. I defined characterising species as those with a phi-coefficient of association >6 or occurring in >70% of plots. A phi-coefficient of >6 was chosen to ensure a minimum of one statistically associated species with each community. Landform and additional vegetation structure information was taken from plot sheets and observational data where available. Geographical distribution came from the final mapping. Where communities were represented by fewer than three plots in analyses I used non-detailed or observational plots for additional information. To describe communities determined by supervised techniques I used species, structure and landform information from non-detailed plots and observational plots, and, where it was diagnostic, mapped distribution. These community descriptions are necessarily less robust but allow indicative recognition in the field.

The ease and certainty with which new plots can be reliably allocated into a classification system outside of an analysis process is important (De Cáceres & Wiser, 2012) and I expected my descriptive-framework to enable this. To test this, I used the 'non-detailed' plots previously excluded from analysis as 'new' plots. I matched the information available from

each plot to that in the descriptive-framework, subjectively assigning it to a vegetation community and rating its level of fit-to-community as high or low. These non-detailed plots had a variety of vegetation information available ranging from a community label with or without a limited species list (and sometimes growth form) to complete species lists with alternative abundance measure such as classes, stem density or basal area and an indication of which layer species occurred in. In plots which had only a label (or label and a species list) I took the label as an indicator of dominance and structure. I also used landform information where it was provided on the site pro-forma.

Along with defining communities in a classification system via consistent analytical techniques, labelling communities using consistent naming conventions is important (De Cáceres & Wiser, 2012). Neldner, Wilson, Dillewaard, Ryan, and Butler (2017) outlines these for the RE framework. In this, a limited number of characterising species are listed in order of dominance, with punctuation to indicate relative abundance and frequency, followed by the structural formation. Associated habitat characteristics, such as landform or soil descriptors are included in labels where they are diagnostic. I followed these conventions to develop community labels.

Results

Plant Communities

There were 57 communities in my study's final classification system, 27 on the Tertiary landscape and 30 on the igneous landscape. Seventy-five percent of these were identified by the new class definition procedures and 25% by supervised techniques and less detailed plot data (Table 4.1). Two communities were recognised after additional analyses requested by the technical review committee (Appendix 4.3). Incorporating my new class definition procedures resulted in fewer communities on both landscapes than the existing supervised classification with an overall reduction of 49%. Individually, the reduction was higher on the Tertiary landscape (54%) than the igneous landscape (42%), driven by the larger decrease in the number of woodlands and shrublands identified (Table 4.1). Whilst most of the final

REs consisted of one plant community, in 11 instances, the review committee assigned several communities to individual REs. The 27 communities on the Tertiary landscape were assigned to 21 REs, and the 30 on the igneous landscape to 23 forming some REs with more than one community (Appendix 4.2). I have not included the detailed descriptions, conservation status and ecological notes for individual REs and their communities as they are available on-line (http://www.qld.gov.au/environment/plants-animals/plants/ecosystems). However, to portray the communities and REs recognised I have included the short label descriptions, mapped areas and notes for the REs in Appendix 4.2. To illustrate the floristic relationships between the communities and REs on each landscape I formed community dendrograms and ordination plots from the detailed plot data (Appendix 4.4).

Table 4.1 The number of communities in each vegetation formation on each landscape. The quantitative analysis resulted in a reduction in the number of vegetation communities. '*a priori*' classification = pre-existing vegetation communities recognized using supervised techniques.

	Grasslands	Shrublands	Woodlands
Tertiary landscape (45 000 km^2)	806 ha	1 904 km^2	46 089 km^2
Quantitatively derived	1	1	17
Qualitatively derived	1	1	6
Total after review *(no. of REs)*	**2** ***(1)***	**2** ***(2)***	**23** ***(18)***
a priori classification	4	7	48
Igneous landscape (5 500 km^2)	154 km^2	110 km^2	5 236 km^2
Quantitatively derived	5	3	16
Qualitatively derived	1	1	4
Total after review *(no. of REs)*	**6** ***(5)***	**4** ***(4)***	**20** ***(14)***
a priori classification	7	6	38

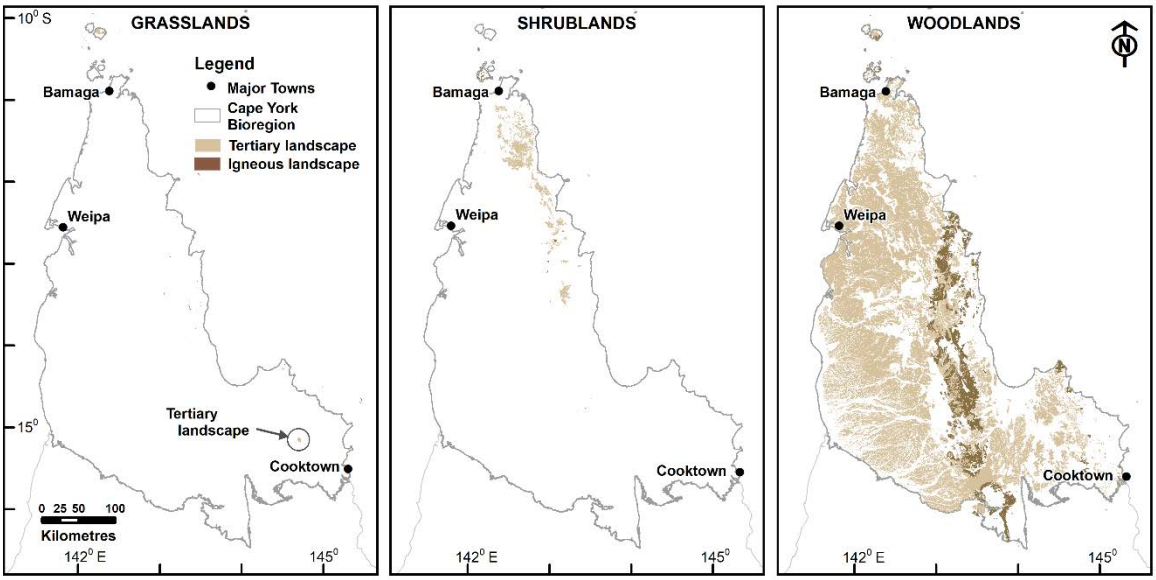

Figure 4.1 Distribution of the vegetation formations across Cape York Peninsula bioregion included in this study.

Summary of plant communities and formations of the Tertiary landscape (old loamy and sandy plains)

Grasslands are of limited extent on the Tertiary landscape (0.01% of the landscape) and contain two communities. One occurs only on islands in the Torres Strait and the other in southern Cape York Peninsula (Figure 4.1). Shrubland communities cover 4% of the landscape (Figure 4.1), the most extensive of which (1 900 km^2) occur on the deep sand plains in the north-east and east of the bioregion. The second occurs only on the Torres Strait islands. Woodlands dominate the Tertiary landscape (95% of the area) (Figure 4.1) and can be broadly categorised into four groups; 1) *Eucalyptus tetrodonta* dominated woodlands, 2) other *Eucalyptus* and *Corymbia* dominated woodlands, 3) *Melaleuca* dominated woodlands, and 4) *Asteromyrtus* dominated woodlands. The *Eucalyptus tetrodonta* woodlands dominate the landscape, covering 42 870 km^2. *Melaleuca* dominated woodlands cover the next largest area of 2 825 km^2, the *Asteromyrtus* dominated woodlands 1 044 km^2 and *Eucalyptus* and *Corymbia* woodlands other than *Eucalyptus tetrodonta* cover the smallest area (528 km^2).

Summary of plant communities and formations of the igneous landscape (hills and lowlands on granitic rocks)

Grasslands are again of limited extent on the igneous landscape, covering 2% (Figure 4.1). The most widespread of these was the rock pavements with scattered herbs and forbs associated with the tops of the major mountain chains on the mainland and the Torres Strait islands (66 km^2). The remaining five are all dominated by Poaceae species. Shrublands cover 12% of the landscape (Figure 4.1), with three of the four communities dominated by *Melaleuca* species. The fourth, covering the largest area (57 km^2), is dominated by an endemic species, *Leptospermum purpurascens*. Despite having the largest area, its range is restricted to the hills and mountains associated with Iron Range in the centre of the bioregion. Woodlands are again the most widespread formation (75% of landscape) (Figure 4.1). These are dominated by *Eucalyptus tetrodonta* woodlands (41% of woodland area) and ironbark woodlands (*Eucalyptus cullenii* and *Eucalyptus crebra*) (28%). Other *Eucalyptus* and *Corymbia* dominated woodlands cover 21%. *Melaleuca* woodlands cover 3% of the landscape, a much smaller area than on the Tertiary landscape. The remaining area is covered by one mixed species low woodland and two *Acacia* communities (both of which occur only in the Torres Strait islands).

Assigning new plots into the classification system

Using the descriptive-framework (Appendix 4.1) I was able to incorporate all 83 non-detailed plots into the classification system. The characterising species provided the most useful information; strength of association allowed us to rank characterising species in importance for a community. The species information in the non-detailed plots could then be matched to this, even when not all characterising species were recorded at a site. While the characterising species was the most useful individual piece of information, the most powerful tool for assigning plots into the classification system was the combination of characterising species plus vegetation structure information. Landform became diagnostic where the characterising species overlapped (particularly the *Eucalyptus tetrodonta* woodlands). I

could assign 66% of plots (55) with a high level-of-fit to community. These were the plots that contained quantitative abundance and structure data collected using different methods. The plots assigned with a low level-of-fit to community were those with only a community label to indicate abundance and structure.

Inclusion of results in mapping

Vegetation mapping and classification systems are two separate processes often accompanying each other (J. Franklin, 2013). In this survey the process was iterative, with the mapping (and accessibility) driving the choice of transects, and the outcomes feeding back to change the supervised classification system depicted in the mapping. Continuing this process, the results of my classification analyses were used to revise the regional ecosystem mapping to reflect the updated vegetation communities and REs. As part of the mapping, individual mapped areas (i.e. polygons) are also assigned levels of reliability for attributes and locational accuracy. Polygons which contained detailed plots were given a high reliability in the mapping as were areas containing non-detailed plots assigned into the classification with a high fit-to-community. Polygons containing non-detailed plots with low fit-to-community were mapped with a low reliability and identified as requiring further survey.

Discussion

I present for the first time, a bioregional scale classification exercise identifying vegetation communities within the RE system which incorporates quantitative analyses. After initial assignment of plots to landscapes, I determined membership of plots to communities using; 1) the new un-supervised class definition procedures proposed in Chapter 3, and 2) statistical analysis of vegetation structure and environmental factors. These communities were incorporated in to the RE classification system by an expert panel peer-review process. I developed a descriptive-framework to characterise the vegetation communities using statistically derived floristic attributes and non-statistically derived abiotic variables. Using this, I assigned new plots into the classification system. In so doing I addressed the two main tasks of a classification system as outlined by De Cáceres and Wiser (2012): to determine

vegetation communities using transparent and repeatable techniques and to provide a framework for consistent and reliable assignment of new plots into the classification system.

While my classification exercise incorporates quantitative analyses as much as available data will allow, 25% of communities were still identified using supervised techniques. This was done using plots with different data collection methods or observational data from helicopter flights over inaccessible areas of the bioregion, meaning the data could not be used in the analyses. Communities identified by supervised techniques only therefore represent ‘known unknowns’ and provide a targeted direction for future data collection.

A notable outcome of applying the new class definition procedures was the 49% reduction in the number of communities recognised compared to the expert-driven process. Quantitative analyses allow experts to test their interpretation of the factors influencing landscape function; in this case, unquantified floristic and biophysical attributes. One question my analyses asks is, ‘Does the floristic composition of the landscape reflect the divisions chosen by experts based on their assumptions about the importance of these attributes?’ The 49% reduction suggests that, in this case, it does not. Quantifying the differences between the supervised and un-supervised communities is considered in Chapter 5. However, one function of quantitative analyses is helping gain consensus among experts about the species driving vegetation community differences.

A major function of a classification system is to allow new plot data to be assigned into it (De Cáceres & Wiser, 2012). In the authors’ experience, an important issue when using a supervised classification system for this task is ambiguity in allocating new plots into the system. A descriptive-framework based on quantitative data helped overcome this by allowing us to allocate plots with different data collection methods into the classification system with a high level-of-fit to community, enhancing the repeatability of allocating new plots. This, in turn, increases the classification system’s applicability by allowing; 1) easier recognition of community types, 2) greater confidence in identifying plots from communities new to the classification system, and 3) the classification system to become a dynamic

system responsive to new information. My descriptive-framework does not fit the definition of membership rules outlined by De Cáceres and Wiser (2012) in that the same rules used to define communities are not used to allocate new plots into it. However, it performs a similar function, thus fulfilling this requirement of a classification system.

A potential benefit of incorporating quantitative analyses into the RE system is in allowing a display of the relationships between communities not obvious in a supervised classification system. An area with a high number of, but similar REs may not support as great a diversity as an area with a lower number of dissimilar REs. For instance, one result of the committee process of allocating communities to REs based on non-floristic variables is that REs can contain communities dominated by different species and with low similarity to each other. Dendrograms, scatter plots and similarity matrices produced by quantitative analyses provide a visualisation and measure of the similarities between REs and their vegetation communities (Appendix 4.4). An example of this is RE 3.12.18 with two communities 'a' and 'b' (Appendix 4.4). In this situation 3.12.18b is found in small patches scattered through larger areas of 3.12.18a, on the same landform, and not predictable enough to be reliably mapped at 1:100,000 scale. Displaying the relationships between communities may be useful, for example in conservation planning.

Conclusion

Globally recognised goals of vegetation science include standardising classification procedures across large geographic areas and multiple administrative boundaries (De Cáceres et al., 2015; Jennings et al., 2009; Walker et al., 2013). These procedures are generally described as standardised data collection methods, classification systems and a quantitatively based classification approach. In Australia, most state governments have adopted approaches which work towards achieving these goals (Gellie, Hunter, Benson, Kirkpatrick, et al., 2018; Sun, Hnatiuk, & Neldner, 1997) and in Queensland this is well advanced. As well as having state-wide RE mapping at 1:100,000 scale, there is a standardised classification system, data collection methods and supervised classification

approach. Extending my quantitatively based classification approach to the RE framework across the remainder of Cape York Peninsula, and other bioregions in Queensland, will further the achievement of these globally recognised goals.

Acknowledgements

This work was carried out with the support of the Queensland Herbarium, Department of Environment and Science, Queensland Government. I thank Peter Bannink for the map figures. I particularly thank the 18 members of the expert panel for their time and commitment in attending the technical review committee workshop in Cairns, Queensland in 2015.

Chapter 5 Supervised versus un-supervised classification: A quantitative comparison of plant communities in savanna vegetation

This chapter is based on a published paper (Eda Addicott & Laurance, 2019), with format and content edits to reduce repetition:

Addicott, E., & Laurance, S. G. W. (2019). Supervised versus un-supervised classification: A quantitative comparison of plant communities in savanna vegetation. *Applied Vegetation Science,* 22, 373–382. doi:10.1111/avsc.12442

Statement of contribution of others:

Addicott conceived the idea, ran all analyses and wrote the manuscript. Laurance provided intellectual assistance and helped write the paper.

Contextual overview

After establishing that my new classification approach is robust at a bioregional scale, I follow this by quantitatively evaluating the differences in the communities identified by the existing supervised approach and my new approach. This is rarely done as most classification exercises which identify plant communities from the same dataset using un-supervised and then supervised plot-grouping techniques have differing concepts and criteria.

Introduction

Plant communities identified by grouping vegetation plot data, forming a classification (Goodall, 1973), underpin many land management decisions and much scientific research (Chytrý et al., 2011; De Cáceres et al., 2015; Jennings et al., 2009). Such classifications allow comparisons of spatial and temporal change across large geographic areas (Jennings et al., 2009; Kent, 2012; Walker et al., 2013) and may underpin environmental legislation (European Commission, 2003; Queensland Government, 1999).

Approaches to grouping vegetation plot data into communities to form a classification can be divided in to three broad types; supervised, semi-supervised and un-supervised (De Cáceres et al., 2015). Un-supervised approaches use statistical and mathematical models (Goodall, 1973; Kent, 2012), semi-supervised use a combination of expert-based knowledge and statistical modelling or formalised rules (Bruelheide, 2000; Tichý et al., 2014) and supervised approaches allocate plot data to communities primarily on the researcher's ecological knowledge and expertise gained through field work observations. Using un-supervised or semi-supervised techniques to recognise communities is recommended to address some of the inconsistencies in many supervised methods such as low repeatability, opaque grouping criteria and their inconsistent application (Kent, 2012; Mucina, 1997). However, supervised methods are still widespread particularly in areas with a long history of vegetation classification (Tichý et al., 2014) and in geographical areas with small numbers of

researchers working in them (Gellie, Hunter, Benson, & McCreery, 2018; Guo et al., 2018; Peet & Roberts, 2013).

Irrespective of the approach used, the final classification needs to be evaluated for its 'appropriateness', dependent on its final use (De Cáceres et al., 2015). Criteria used for this are either internal or external (Gauch & Whittaker, 1981). Internal criteria utilise attributes of the community composition data, and external criteria focus on abiotic and biotic attributes not contained within the data, such as environmental variables or ecological interpretability by experts (De Cáceres et al., 2015). No matter the evaluation criteria used, communities derived using un-supervised or semi-supervised methods are commonly evaluated in a separate process to their formation, while supervised methods may combine the two processes (De Cáceres & Wiser, 2012). Evaluating any classification for ecological interpretability is common practice and often done through the process of peer-review (Kent, 2012).

Comparing classifications derived from supervised and un-supervised approaches is problematic. They usually use different primary vegetation attributes, and supervised methods often do not have explicit classification criteria (De Cáceres et al., 2015). Hence comparisons of these classifications are limited to external criteria and expert evaluation of the floristic community concepts recognised (for example Knollová et al. (2005); Koci et al. (2003); Tichý et al. (2014); Wiser et al. (2011). In north-eastern Australia two classifications of communities at the plant association level are available. One classification is derived using supervised methods (supervised classification system) and the other using un-supervised methods (un-supervised classification system). The classification systems fit into the standardised classification framework used by the government of Queensland, a state of Australia (Sattler & Williams, 1999). Both classification systems cover two landscapes with different environmental gradients. They use the same vegetation dataset, and importantly, the same a priori classification criteria, to form communities based on comparable primary vegetation attributes. This provides the rare opportunity to use evaluation criteria based on

community composition data to directly compare the characteristics of communities recognised using these two common approaches. Specifically, in this study I compare and contrast communities identified between the two classification approaches and landscapes using criteria based on the recognisability of communities and their usefulness as land management tools (Addicott, Laurance, et al., 2018; De Cáceres et al., 2015; Roberts, 2015; Tichý, 2002).

Methods

Study area

My study area in this chapter is the same as Chapter 4, the Tertiary and igneous landscapes of CYP (Figure 1.2). The igneous landscape is, on average, significantly cooler, higher, and steeper, with more seasonal variation in temperature and higher soil moisture in drought periods than the Tertiary landscape (Figure 5.1, Appendix 5.1). The range between minimum and maximum for these variables is, however, similar between landscapes, except for altitude and slope, where the igneous landscape has greater range than the Tertiary landscape (Figure 5.1).

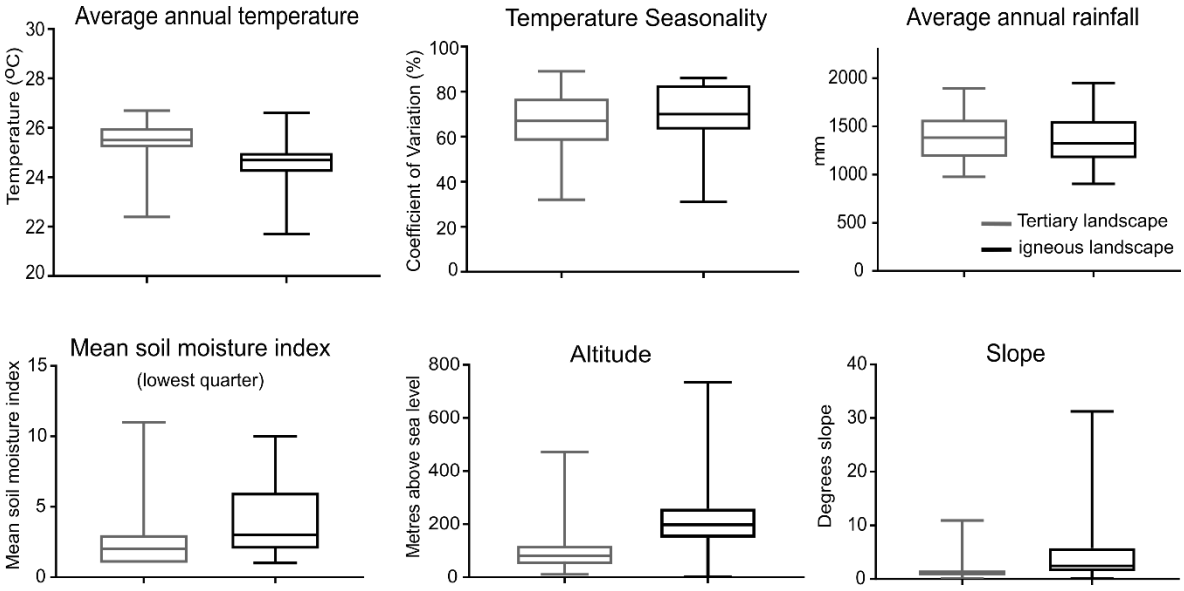

Figure 5.1 Differences in environmental variables on the Tertiary and igneous landscapes. The igneous landscapes are, on average, significantly cooler, wetter, higher and steeper, but

with more soil moisture in drought periods and greater seasonal variation in temperature. The igneous landscape also has a greater range in slope and altitude.

Dataset

I used the same dataset of 288 plots as chapter 4 with 192 plots and 241 species on the Tertiary landscape and 96 plots and 258 species on the igneous landscape.

Forming classifications

Both classification systems were formed independently using the same plot data and *a priori* classification criteria, based on attributes used to differentiate communities for mapping at landscape scales (Neldner, Wilson, et al., 2017). These criteria specify recognition of communities based on the %PFC of dominant and sub-dominant species in each vegetation layer. Initial recognition is based on the predominant layer, with further differentiation based on a floristically consistent sub-canopy, shrub or ground layer (Appendix 1.1). A group of three experts applied these criteria to the plot data in an unquantified, subjective process to form the supervised classification (Regional Ecosystem Description Database (REDD) Version 9.0. Queensland Department of Science, Information Technology and Innovation, unpublished). In Chapter 3 (Addicott, Laurance, et al., 2018) I quantified these criteria by removing sparse species based on their percent contribution to the total foliage cover at a site (identifying dominant and sub-dominant species) and multiplying %PFC by the height of the vegetation layer (identifying the pre-dominant layer and consistent sub-canopy or shrub layers). In Chapter 4 ((Addicott, Newton, et al., 2018) I applied this to form the un-supervised classification in my comparison using the Bray-Curtis coefficient and agglomerative hierarchical clustering (unweighted pair group mean averaging), then Indicator Species Analysis (Dufrêne & Legendre, 1997) to decide on clusters to use as plant communities. The communities in each classification were therefore formed using the same primary vegetation attributes of dominant species in the predominant layer with consideration given to the lower vegetation layers (De Cáceres et al., 2015). Importantly, both classification systems in this study were reviewed by the same experts for ecological interpretability using a consistent set

of external criteria; landform, geology, observational field data, soils and geographical distribution. The communities in the supervised classification system were formed and reviewed for ecological interpretability in the same process, while the communities in the un-supervised classification system were reviewed for ecological interpretability in a separate process.

Distinguishing communities based on a different landform and / or geological substrate, or on different dominant species in the ground layer are also included as part of the a priori classification criteria (Appendix 1.1). After careful examination I found none of the supervised communities were based solely on a difference in landform or dominant species in the ground layer, with communities on different landforms all having accompanying compositional or structural differences. The classification systems are therefore methodologically consistent, differing only in their class definition procedures. The plant communities can therefore be meaningfully compared using evaluation criteria based on community composition data.

Evaluating the differences between communities identified by different methods and on different landscapes

A number of criteria using primary vegetation attributes for evaluating classification systems are available (De Cáceres et al., 2015). I utilised four based on the recognisability of communities and their usefulness for land management purposes. I evaluated the recognisability of communities by: 1) distinctiveness of communities, meaning they have low similarity in species composition to others (De Cáceres et al., 2015), and 2) identifiability of communities, measured by the degree to which characterising species overlap between communities (Chytrý & Tichý, 2003). To evaluate the communities' usefulness for land management I used: 3) internal community-heterogeneity, indicating a consistent level of division within the classification (at the plant association level for instance) (Ferrier, 2002; Lawson et al., 2010), and 4) predictability of species distribution across the classification

(Lyons et al., 2016), allowing a more accurate prediction of species occurrence in the landscape.

All calculations were done using JUICE software (version 7.0.102,(Tichý, 2002)), PRIMER 7 software (version 7.0; PRIMER-e, Quest Research Limited) or R software (version 3.4.0; www.r-project.org, package 'optimus'). Mann-Whitney U tests and t-tests were done using GraphPad Prism 7 for Windows (version 7.04; GraphPad software Inc., La Jolla California USA).

Communities are more distinctive if they have a low similarity in species composition to others. This can be measured by how much overlap there is in the similarity of sites within versus between communities (De Cáceres et al., 2015). To do this I used the ANOSIM algorithm (K. R. Clarke & Green, 1988) which computes an R statistic for every pair of communities, with the R value reflecting the difference in similarity of sites within a community, compared to between two communities. If the sites in a community have no similarity to any sites in another community then R = 1. The more similar sites are between communities the closer the R statistic is to 0. If all sites in a community are more similar to sites in another community the R value = -1. I compared the proportional distribution of R values across their range (-1 to +1) in each classification by grouping them into three 'R-groups': 1) 'not-distinctive', R = -1 – 0.4 (sites in one community are as similar to sites in another community); 2) 'distinctive', R = 0.41 – 0.8 (most sites in a community are more similar to each other than to sites in another community); 3) 'very-distinctive', R = 0.81 – 1 (all sites in a community are more similar to each other than to sites in another community) (P. J. Clarke et al., 2014). I tested for changes in proportions of R statistics between not-distinctive, distinctive and very-distinctive groups using a chi-squared test.

There are two indices available for measuring identifiability which use the strength of association of characterising species to communities. The 'Sharpness' index is the proportion of a community's characterising species relative to the average species richness of its sites, and values range between zero and infinity (Chytrý & Tichý, 2003). The

‘Uniqueness’ index of a community is measured by the proportion of a community’s characterising species not shared with other communities. Values for Uniqueness range between zero (all of a community’s characterising species are shared with other units) and one (none of a community’s characterising species are shared with other units) (Chytrý & Tichý, 2003). Where a community has no characterising species Uniqueness is undefined (Tichý, 2002). For the purposes of my calculations I gave these a value of zero. I determined characterising species for communities using the Φ-coefficient (Chytrý et al., 2002). The un-supervised classification system used statistically significantly associated species ($p<0.05$) with a Φ-coefficient >6 as characterising species (Addicott, Newton, et al., 2018) and I adopt the same threshold for the supervised classification. I tested for difference between the average Sharpness of classifications using an unpaired t-test on the igneous landscape but with a Mann-Whitney U test on the Tertiary landscape as these values had a non-normal distribution. For differences in average Uniqueness I used an unpaired t-test for both landscapes.

Classifications whose communities have similar levels of internal heterogeneity are more useful for conservation planning (Ferrier, 2002). To measure this, I used the average similarity of sites within a community (using the SIMPROF algorithm, (K. R. Clarke et al., 2008)) as a measure of the compositional variability of a community (within-community similarity) (De Cáceres et al., 2015). The lower the average similarity the more variable the community’s composition. I compared the variability of within-community similarity between classifications using the maximum - minimum range, standard deviation and interquartile range.

A classification system which better predicts species foliage cover among its communities will be better able to predict the distribution of species across the landscape (Lyons et al., 2016), hence making it more useful for land management (Margules & Pressey, 2000). To measure this M. B. Lyons et al. (2016) utilise generalised linear models (GLMs) and the Akaike’s Information Criterion (AIC), in which the AIC is summed across individual species in

each classification system. This represents a “goodness of fit” measure. The lower the sum-of-AIC the better a classification system predicts the distribution of species foliage cover in the dataset. I utilised the delta-AIC score for individual species to assess the difference between the classifications. A species’ delta-AIC of >4 is considered as a substantial change (as opposed to statistically significant) (Murtaugh, 2014). Therefore, if the difference in the sum-of-AIC scores between classification systems is greater than four multiplied by the number of species in the dataset, a classification can be regarded as substantially better at predicting the distribution of species (Murtaugh, 2014). For the difference between classifications to be substantial on the Tertiary landscape I needed a difference in sum-of-AIC scores >956 and on the igneous landscape a difference >1032. These methods are available in the R package ‘optimus’.

When comparing different classification stems an obvious question to ask is whether they identify the same communities. There are two aspects to this: the similarity of the central floristic concepts, and the similarity of the compositional boundaries between them (De Cáceres et al., 2015; Tichý et al., 2014). To compare the central concepts, I assumed that groups with highly similar characterising species described the same communities (Knollová et al., 2005). I formed synoptic tables for each classification (Appendix 5.4) and used the ‘compare two synoptic tables’ routine in the JUICE software to calculate a percent-similarity of central concepts between the classification systems. This routine calculates a table of Euclidian distances between all pairs of synoptic columns in two synoptic tables to find their total similarity (Knollová et al., 2005). Similarity matrices formed from community species-data also provided information, not only about the similarity of the floristic concepts, but also the similarity of compositional boundaries between the central concepts. The similarity matrices use all species in the dataset, whereas the synoptic table comparison is based only on the characterising species. If both the synoptic tables and the similarity matrices had high levels of agreement, then both floristic concepts and compositional boundaries would be similar between methods. I firstly formed community species-data by averaging site species-

data (Gauch & Whittaker, 1981), then calculated a community similarity matrix for each classification system (with the Bray-Curtis similarity coefficient). I used the 2STAGE routine in the PRIMER-e software and calculated a Spearman rank correlation between the two matrices. This routine calculates the correlation between the elements of two similarity matrices to provide a summary of the extent of 'agreement' between the representations of the multivariate data of two matrices.

My results in testing for the same communities led me to additionally investigate differences in the classification systems between landscapes as they have different environmental gradients. I evaluated differences using identifiability and distinctiveness and tested whether the similarity between communities was different on each landscape. To test the latter, I used the average within-community similarity and a Mann-Whitney U test.

Results

Un-supervised methods identified fewer plant communities than supervised methods. For example, across 48 000 km^2 of savanna vegetation on the Tertiary landscape supervised methods recognised a total of 37 communities, while un-supervised methods recognised 18. On the igneous rock landscape (5 500 km^2) supervised methods recognised 37 communities and un-supervised methods recognised 25 communities (Table 5.1).

Table 5.1 Number of communities in each classification on each landscape, their average Sharpness and Uniqueness, and the sum-of-AIC scores. The lower the sum-of-AIC the better a classification predicts the distribution of species foliage cover within the dataset. Significance levels indicated by * = p <0.05, ** = p <0.01, *** substantial difference in sum-of-AIC scores.

	Tertiary landscape *(n=192)*		Igneous landscape *(n=96)*	
Method	supervised	un-supervised	supervised	un-supervised
Total number communities	37	18	37	25
Number of communities represented by 1 site	11	3	12	7
Average Sharpness	107	161*	167	226*

Average Uniqueness	0.4	0.7**	0.6	0.8**
Sum-of-AIC	5037	-2588***	12 609	5495***

Differences between methods

Distinctiveness

Plant communities recognised using un-supervised methods were significantly more distinctive than those recognised using supervised methods. Although small, the increase in the proportion of values in the 'distinctive' and 'very-distinctive' R-groups between methods was significant on both the igneous ($\chi^2(3, N = 879) = 18.9$, $p < 0.001$) and Tertiary landscapes ($\chi^2(3, N = 761) = 10.7$, $p = 0.01$). On the Tertiary landscapes 78% of communities in the supervised classification system were very-distinctive, increasing to 83% in the un-supervised classification. On the igneous landscapes 89% of supervised communities were very-distinctive increasing to 97% in the un-supervised classification system (Table 5.2).

Table 5.2 Proportions of R values indicating the proportion of communities that are distinctive on each landscape using the supervised and un-supervised analysis method. On both landscapes, un-supervised methods recognised communities that were more distinctive from each other than supervised methods. Significance indicated by * p <= 0.01

	Number of R values in each group *(%)*		
	Not-distinctive	Distinctive	Very-distinctive
R range	*-1 - 0.41*	*0.41 - 0.8*	*0.81 - 1*
Tertiary landscape			
supervised methods	55 *(9%)*	79 *(13%)*	477 *(78%)*
un-supervised methods	2 *(1%)*	*24 *(16%)*	*124 *(83%)*
Igneous landscape			

supervised methods	29 *(5%)*	37 *(6%)*	534 *(89%)*
un-supervised methods	0	*8 *(3%)*	*271 *(97%)*

Identifiability

As in distinctiveness, the identifiability of communities in the un-supervised classification system was significantly greater than those in the supervised classification system. Sharpness was significantly higher on both landscapes (Tertiary: Medians (supervised = 96.9, un-supervised = 135.4) Mann-Whitney U = 216, p = 0.03; igneous: $t_{(59)} = 2.50$, p = 0.02) indicating that a greater proportion of species in the un-supervised communities were characterising species (Table 5.1, Appendix 5.2). Un-supervised methods also resulted in communities with significantly higher Uniqueness indices on both landscapes (Tertiary: $t_{(53)} = 3.04$, $p<0.01$; igneous: $t_{(59)} = 3.93$, $p<0.01$) meaning there was less sharing of characterising species across communities in the un-supervised classification system than the supervised system (Table 5.1, Appendix 5.2).

Internal community heterogeneity

There were more even levels of internal-heterogeneity in the supervised classification as evidenced by the lower variability of within-community similarity. There was a reduction of the interquartile range and standard deviation and no change in the max-min range on the igneous landscape while on the Tertiary landscape there was a slight increase in the interquartile range, but a reduction in the max-min range and standard deviation (Figure 5.2, Appendix 5.3). An additional, interesting, outcome was that the average within-community similarity was the same in both classification systems (Figure 5.2, Appendix 5.3). This was surprising as the supervised classification system had more communities, more of which were represented by single sites (Table 5.1) and therefore not included in the average-similarity calculations. With fewer sites spread across more communities and therefore less

sites per community, I expected less floristic variability in the supervised communities with a correspondingly higher within-community similarity.

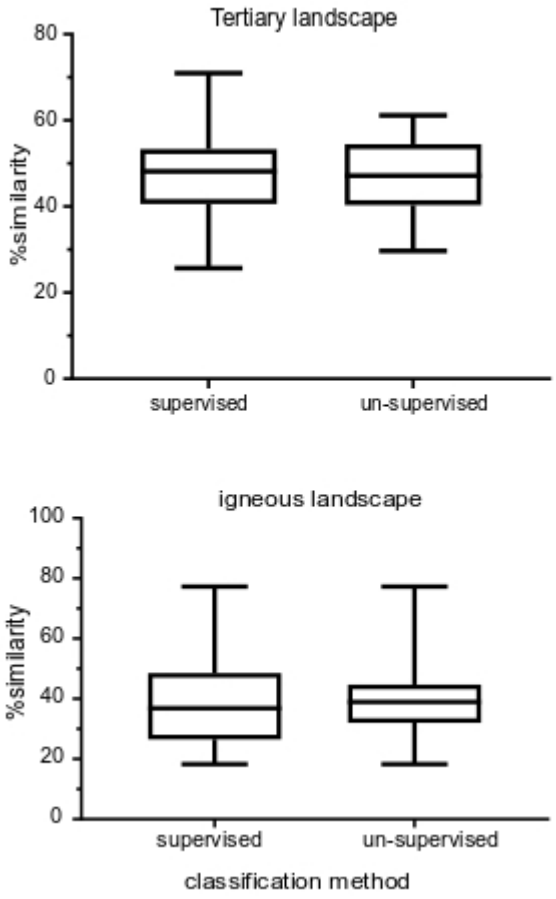

Figure 5.2 Variability in %similarity of sites within a community in the supervised and un-supervised classification. Greater variability represents more internal heterogeneity within communities.

Prediction of species foliage cover

The un-supervised classification system was substantially better able to predict the distribution of species foliage cover among communities. The difference in the sum-of-AIC score was >7000 on both landscapes, considerably higher than the differences of 956 on the Tertiary landscape and 1032 on the igneous landscape required to be substantial (Table 5.1).

Similarity of community concepts and compositional boundaries

Both methods recognised similar floristic community concepts, but the recognition of the compositional boundaries between them differed between landscapes. The floristic concepts on the igneous landscape were 90% similar between methods. Despite having half the number of communities in the un-supervised classification system, the floristic concepts on the Tertiary landscape were 78.5% similar. This was because the supervised methods identified communities with very similar floristic concepts to each other, with each of these being similar to one community identified by un-supervised methods. The correlation between the community similarity matrices between methods was strong on the igneous landscape (rho = 0.9). However, on the Tertiary landscape the correlation was rho = 0.5, indicating that while the central floristic concepts of communities were similar, the similarity of compositional boundaries of communities were no better than random.

Differences between landscapes

Identifiability and distinctiveness of communities on the igneous landscape, with its steeper environmental gradients, were higher than the Tertiary landscape. This was irrespective of the method used to recognise them. Sharpness and Uniqueness (measuring identifiability) were significantly higher on the igneous landscapes in the supervised and un-supervised classification systems (supervised: Sharpness (Medians igneous = 146.7, Tertiary = 96.87 Mann-Whitney U = 390, $p<0.01$); Uniqueness $t_{(72)} = 2.35$, $p=0.02$), (un-supervised: Sharpness $t_{(40)} = 2.28$, $p=0.03$; Uniqueness $t_{(40)} = 2.27$, $p=0.03$) (Table 5.1, Appendix 5.2). The proportion of communities that were very-distinctive, measured by the proportion of R values close to 1, was also significantly higher on igneous landscapes in both classification systems (supervised, $p<0.0001$; un-supervised, $p<0.0001$) (Table 5.2). The similarity of communities to each other on the Tertiary landscape, with its more gradual environmental gradients, was significantly greater than those on the igneous landscape (Medians (igneous = 8.5, Tertiary = 22), Mann-Whitney U = 9640, $p<0.0001$) (Figure 5.2, Appendix 5.3)

Discussion

Comparisons between supervised and un-supervised vegetation classification systems are common but generally restricted to comparison of central floristic concepts and expert evaluation using external criteria (Gégout & Coudun, 2012; Koci et al., 2003; Neldner & Howitt, 1991; Tichý et al., 2014). In contrast, I compared the compositional attributes of plant communities identified by these two approaches. Across a 53 500 km^2 landscape of mountain ranges, plateaus and plains in northern Australia, I found that the identification of plant communities differed significantly between supervised and un-supervised approaches. Although similar central floristic concepts were recognised as communities in both landscapes, communities identified using supervised methods were significantly less recognisable and less useful for land-use planning on both landscapes. Additionally, the two methods recognised different boundaries between communities depending on landscape. On the landscape with steeper environmental gradients the correlation between community boundaries recognised by each method was strong, while on the landscape with gradual environmental gradients it was no better than random. To a large extent my results are not unexpected. The importance of my work is in quantifying these levels of improvement and giving a greater understanding of the implications of using supervised methods to identify communities.

Differentiating communities is often based on abundance (De Cáceres et al., 2018) and may be described, as in this study, by unquantified concepts of dominant and sub-dominant species (Executive Steering Committee for Australian Vegetation Information & Department of the Environment and Heritage, 2003; Gellie, Hunter, Benson, Kirkpatrick, et al., 2018; Guo et al., 2018; MacKenzie & Meidinger, 2018). The communities from un-supervised analysis had higher proportions of characterising species (Sharpness) which were less likely to be shared across communities (Uniqueness) because un-supervised analysis can consistently apply abundance thresholds, quantifying dominant and sub-dominant. The communities also

had more consistent internal levels of heterogeneity. These improvements will lead to less ambiguous community descriptions, improving recognition in the field and the ability for new sites to be added into the classification system. This, along with the better ability to predict species distribution across the landscape means the classification system is more useful for end-uses such as land managers.

Despite the significant improvement in recognisability of communities derived by un-supervised methods it is interesting to note there was a high level of agreement in the central floristic concepts between the two approaches. A review of current literature found varying levels of similarity between approaches (Douda et al., 2015; Knollová et al., 2005; Tichý et al., 2014; Wesche & von Wehrden, 2011). For example, one of the studies with the highest level of agreement showed 75% of the original supervised detected communities could be numerically reproduced in terrestrial vegetation of Catalonia, Spain (De Cáceres et al., 2009). My study shows higher levels than these (78.5 and 90% on the two landscapes) and I suggest a major reason for this result is the use of the same classification criteria in both approaches (Appendix 1.1,(Neldner, Wilson, et al., 2019)). Criteria assist both approaches to recognise associations reflecting the same scale of environmental gradients. Large scale gradients, such as climate, breakdown at the micro-level and local factors such as competition and disturbance history influence floristic assemblages. If the same classification protocols are not followed by both methods, it is possible the un-supervised methods will identify communities at scales below those of the supervised methods. This may easily occur, for example, when full floristics are used in un-supervised classification exercises, but experts are focusing on dominants or woody vegetation to determine communities at landscape scales (Bedward et al., 1992; Guo et al., 2018; Neldner & Howitt, 1991). Using specific classification criteria has the added advantage of improving the transparency of species' weightings used by experts, one of the major drawbacks of supervised methods (Mucina, 1997). Semi-supervised classification approaches also formalise the central floristic concepts and compositional boundaries of communities to a

greater extent than supervised approaches (Bruelheide, 2000; Landucci et al., 2015; Tichý et al., 2014), and often provide accompanying rules for assigning new sites in to the classification system (Bruelheide & Chytrý, 2000).

It is generally accepted that landscapes with steeper environmental gradients have more distinct communities (van der Maarel & Franklin, 2013). My results support this and show a strong correlation between the compositional boundaries of the supervised and un-supervised communities on the igneous landscape (steeper gradients), and a contrastingly low correlation on the landscape with gradual environmental gradients (the Tertiary plains). This was probably inevitable considering the communities on the Tertiary plains were more similar to each other. They had lower proportions of characterising species, more frequently occurring in more than one community, implying continuous variation across the vegetation (De Cáceres et al., 2010). However, another key factor contributing to the low correlation in these landscapes is likely to be one of the psychological issues surrounding the use of supervised methods. Repeated exposure to similar events promotes familiarity, and exposure to something different triggers cognitive arousal, meaning that brains are hard-wired to register difference (Kahneman, 2011). This contrasts with un-supervised classification methods which search for similarity. Where the boundaries between communities are sharp, such as landscapes with strong zonation or low species richness (Landucci et al., 2015), similarity and differences are more easily recognisable. In broad landscapes with similar floristic assemblages, experts are more likely to be cognitively biased towards attaching meaning to perceived floristic differences. Thus, the findings from my comparison between the supervised and un-supervised communities can be generalised to supervised versus un-supervised or semi-supervised approaches. I posit that any un-supervised or semi-supervised classification approach will result in more recognisable and distinct communities than supervised approaches when applied to the same data. This would include other data measures such as stem density, basal area or presence data. The same factors will be at play wherever a decision is to be made on how to form groups.

Many areas across the savanna biome have small numbers of available researchers. For example, across an area of approximately 260 000 km^2, which includes my study area, there are three researchers in the field of vegetation classification. Australia is a developed country, so available resources may be even more limited in countries with less history of vegetation classification and the prevalence of supervised approaches in identifying communities will be correspondingly high (Peet & Roberts, 2013). Moreover, most vegetation classification systems are hierarchical with plant associations sitting underneath broader vegetation types commonly identified using supervised methods to some degree (De Cáceres et al., 2018). More work is needed to understand the scale of environmental gradients and the levels of classification hierarchies which start to produce unacceptable differences in the concordance of communities between the methods. This would give a greater understanding of the reliability associated with supervised approaches as it is possible the scale of differences delineated by groups higher up the classification hierarchy concord more with the differences observed by experts.

Conclusions

Vegetation classification systems are widely used in landscape management and understanding their limitations is important in their application. The use of supervised approaches for identifying communities is common and my results highlight the importance of understanding the implications of using these approaches. Whilst it is recommended that un-supervised or semi-supervised approaches are used when possible, managers and researchers can have some confidence that supervised approaches may identify the central floristic concepts of communities consistently, especially with the application of a consistent set of classification criteria. However, they need to be aware that communities from supervised methods may be less recognisable and less useful for land management planning and, on landscapes with gradual environmental gradients, it may be more difficult to consistently identify compositional boundaries between communities. If limited resources for un-supervised analysis are available, I suggest targeting broad landscapes with low

environmental variability in the comfort of knowing that on landscapes with steeper gradients, supervised and un-supervised approaches are more likely to identify similar communities.

Acknowledgements

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Chapter 6 Applying the new classification approach in an ecological context: the inter-tidal plant communities in north-eastern Australia, their relative role in carbon sequestration and vulnerability to extreme climate events

This chapter is based on a paper accepted for publication:

Addicott, E., Laurance, S. G. W., Bannink, P. & Thompson, S. Inter-tidal plant communities in north-eastern Australia, their relative role in carbon sequestration and vulnerability to extreme climate events: Baseline conservation information. Accepted in the journal *Aquatic Conservation: Freshwater and Marine Ecosystems*

Statement of contribution of others:

Addicott conceived the idea, ran all analyses (except as mentioned below) and wrote the manuscript. Laurance provided intellectual and methods assistance and helped write the paper. Bannink conducted the GIS analyses and commented on the manuscript and Thompson provided information regarding non-mangrove dieback and commented on the manuscript.

Contextual overview

One of the major functions of classification systems is as tools to manage and understand landscapes. Having determined, tested and evaluated plant communities identified using my new classification approach my next step is to apply it in an ecological context. I chose the inter-tidal communities of CYP as they are recognised globally for providing important ecosystem services such as C sequestration and in mitigating the impacts of climate change. The inter-tidal communities are extensive along the 7 400 km coastline CYP. This chapter provides baseline information on their floristic composition, C sequestration, and vulnerability to climate extremes with which to underpin their effective conservation management. This chapter has been accepted in the journal *Aquatic Conservation: Freshwater and Marine Ecosystems* and is therefore written in the third person to comply with the journal formatting requirements.

Introduction

Mangrove forests and saltmarshes are recognized globally as providing important ecosystem services ranging from uses such as timber harvesting and fuelwood, to habitats such as fisheries nurseries (Barbier et al., 2011; Feller et al., 2010; Himes-Cornell, Pendleton, & Atiyah, 2018). These ecosystems are additionally important in mitigating impacts from the effects of climate change such as wave action, storm surges, flood waters and atmospheric carbon increase (Donato et al., 2011; Doughty et al., 2016). Mangrove forests and saltmarshes also have among the highest carbon stores of global ecosystems (Ewers Lewis, Carnell, Sanderman, Baldock, & Macreadie, 2018). However, they are understood to be severely threatened by anthropogenic factors such as land-use conversion, over-utilization of the services they supply and climate change (Murray, Pendleton, Jenkins, & Sifleet, 2011; Rogers et al., 2016; Sarker, Reeve, Paul, & Matthiopoulos, 2019; Ward, Friess, Day, & MacKenzie, 2016). Effective management of these ecosystems is important for maintaining their ecosystem services. For this to occur,

baseline information is required on their floristic composition, structural diversity and extent (Barbier et al., 2011; Kuenzer, Bluemel, Gebhardt, Quoc, & Dech, 2011; Murray et al., 2011; Rogers, Macreadie, Kelleway, & Saintilan, 2018). Data on vegetation cover, the description, and the extent of these communities are often gathered by satellite imagery classification (for example Hamilton and Friess (2018) due to problems of accessibility (Kuenzer et al., 2011). However, satellite imagery classifications are ideally supported by plot-based classifications derived from ground truthing data (Aslan, Rahman, Warren, & Robeson, 2016; Kelleway et al., 2017).

Mangrove forests and saltmarshes are extensive in Australia, with Australia estimated as having between the 2nd (Giri et al., 2011) to 5th (Hamilton & Friess, 2018) largest area of mangrove forest and some of the major areas of saltmarsh in the world (McOwen et al., 2017). Mangrove forests in Australia occur predominantly around the northern Australian coastline and those in north-eastern Australia are recognized as being of particularly high diversity, comparable to those in Indonesia and Bangladesh (Duke, 2006; Ragavan et al., 2015; Sarker et al., 2019). Currently, the most common threatening processes such as land-conversion and over-utilization are not high in northern Australia due to its sparse population and low development pressure. However, during the El-Nino event of 2015 – 2016. Duke et al. (2017) reported extensive mangrove forest loss around the coast of the Gulf of Carpentaria, in north-eastern Australia, due to associated extreme environmental warming and drying. Northern Australia is also seen as an area of future economic development (Department of the Prime Minister and Cabinet, 2015) with consequent potential damage to the intertidal communities of the area.

The C store of the intertidal communities in Australia has been investigated in individual estuaries around the coastline (D. R. Brown et al., 2016; Ewers Lewis et al., 2018; Owers, Rogers, & Woodroffe, 2018). In northern Australia the C stored in mangrove forests in three estuaries has been quantified using figures calculated from site-specific surveys; Mangrove Bay on the north-west coast (Hickey, Callow, Phinn, Lovelock, & Duarte, 2018), Darwin

Harbour on the north coast (Brocklehurst & Edmeades, 1996) and the Hinchinbrook channel on the north-east coast (Sanders et al., 2016). However, these studies do not reflect the differences in C stored in 'estuarine / deltaic' versus 'oceanic / fringing' habitats (Donato et al., 2011), both of which occur in northern Australia. Information on the C sequestration capacity of mangrove forests in northern Australia is available in one location in the Herbert River estuary (Brunskill, Zagorskis, & Pfitzner, 2002). Data on the C store and sequestration capacity of saltmarshes in northern Australia are only available for one location, also the Darwin Harbour (Macreadie et al., 2017). To date there has been no estimate of the C store and sequestration capacity of the intertidal communities of northern Australia at a regional scale.

Cape York Peninsula bioregion (CYP) is the most north-eastern in Australia (Thackway & Cresswell, 1995) with a coastline of 7 480km, approximately as long as that of Italy. The intertidal communities along this coastline have been classified and described previously, however this was done using qualitative assessment methods (Danaher, 1995; Neldner & Clarkson, 1995). This study has three aims; firstly, to determine the floristic composition of the intertidal communities of the region using quantitative classification methods, secondly to estimate their potential C store and sequestration capacity, and lastly to evaluate the vulnerability of the mangrove forests to future extreme climate events such as the warming and drying associated with the El Nino event of 2015-16.

Methods

Definition of terms and proscription of communities in this study

Intertidal plant communities included in this study are those occurring in the intertidal zone. This is defined as areas inundated daily up to an occasional one or two spring tides per year but excludes areas only subject to storm surges above normal tides (Wilson & Taylor, 2012). This zone includes two vegetation formations, mangrove forests and saltmarshes, but excludes seagrass communities. Mangrove forests are forests dominated by trees, shrubs,

palms or ground ferns taller than one half metre and which normally grow in the intertidal zone (Duke, 2006). The term ‘mangrove forest’ refers to the habitat and ‘mangrove’ refers to the tree species which occur in the intertidal zone. This study distinguishes between ‘true’ mangroves and mangrove associates using the definitions of Tomlinson (2016). Saltmarshes include communities dominated by herbs, grasses, low shrubs and salt flats (Saintilan, 2009b).

Study Area

The coastline of CYP extends over 7 480 km in north-eastern Australia. Situated in the monsoon tropics, CYP has distinct wet and dry seasons. A short summer wet season (December to March) is characterised by extensive flooding and run-off events (Bardsley, Davie, & Woodroffe, 1985) contrasting with a low run-off dry season (June - November). These contrasts in freshwater input result in fluctuating levels of salinity in the intertidal zone throughout the year (Duke, 2006). CYP also experiences infrequent tropical cyclones and storm surges (Bardsley et al., 1985). Tide heights around the coast range from three to five metres (http://www.msq.qld.gov.au/Tides/Tidal-datum-information accessed on 2/11/16). The bioregion has both a west-facing coastline that forms the east coast of the Gulf of Carpentaria and extends south into the Gulf Plains, and an eastern coastline that connects with the Wet Tropics bioregion (Figure 6.1). These coastlines differ climatically and geomorphologically. The east coast spans humid to wet tropical climate zones (Duke, 2006), has numerous bays containing distinct patches of mangrove forests and offshore islands. It is fringed by the Great Barrier Reef lagoon along its extent (Figure 6.1). The only large area of saltmarsh (~ 22 000 ha) on this coastline is on an extensive coastal plain in the southeast (Princess Charlotte Bay, Figure 6.1). The west coast borders the Gulf of Carpentaria and spans arid, semi-arid and humid climate zones (Duke, 2006). There are three major bays where mangrove forests are prominent, while elsewhere mangroves fringe the coastline and tidal inlets. Saltmarshes are prominent along the south-west coastline.

The intertidal communities of CYP are situated in the Indo-West Pacific region of the world's mangroves, straddling the north-western and north-eastern mangrove subregions of Australia (Duke, 1992). They cover approximately 2 604 km^2 (Accad, Neldner, et al., 2019) around the coastline of CYP, extending up to 40 km inland (Figure 6.1) and form part of the continuous, but disjunct, distribution of estuarine ecosystems around the Australian coastline.

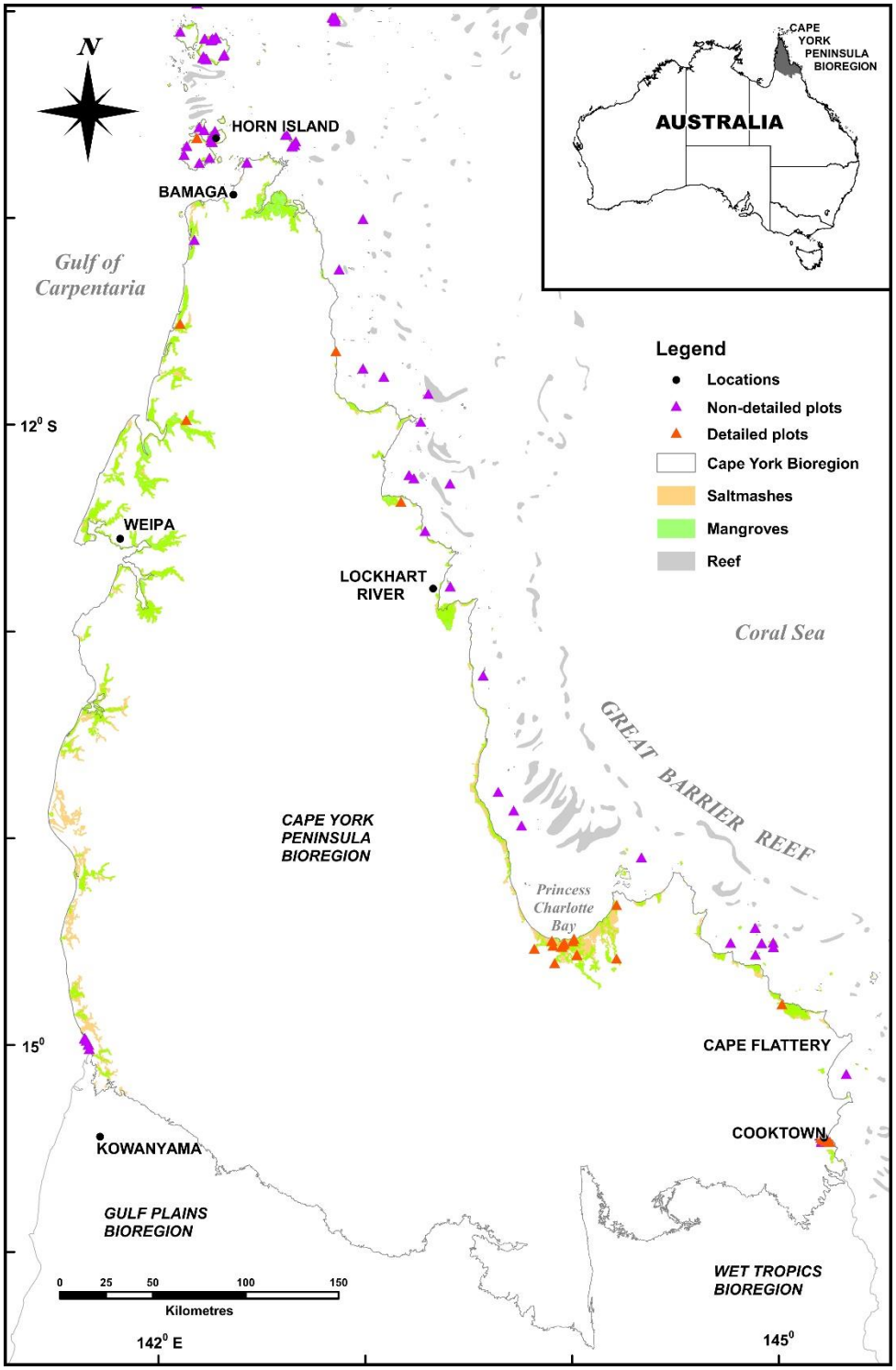

Figure 6.1 Cape York Peninsula bioregion, north-eastern Australia. The Gulf of Carpentaria is on the western side and the Great Barrier Reef fringes the east. The semi-arid Gulf Plains

bioregion lies to the south and west, while the wet-humid bioregion of the Wet Tropics is to the south and east.

Data collation

Vegetation plot data were extracted from the Queensland government 'CORVEG' database and grouped into two categories depending on the level of detail of the data: plots containing data appropriate for use in this quantitative analysis (termed detailed plots) and plots excluded from the analysis because they contained either insufficient data, data collected using different methods or data collected after the completion of this project (termed non-detailed plots). There were 41 detailed plots (predominantly on the mainland) and 182 non-detailed plots, (predominantly on islands) (Figure 6.1). The majority of detailed plots were collected between 1992 and 1996 as part of a comprehensive vegetation survey and mapping project (Neldner & Clarkson, 1995) with data collected using the standardized, state-wide survey and mapping methodology of Queensland (Neldner, Wilson, et al., 2019). Within plots of 50 m x 10 m all woody vegetation abundance was measured in three ways; 1) percentage projected foliage cover (%PFC) for each species in each woody vegetation layer along a 50m transect using the line intercept method, 2) stem count of each species in each layer in the plot, and 3) basal area of each species in each layer. The maximum, minimum and average height of each layer was also recorded. The ground layer had species' abundance recorded as an average %PFC of five x 1 m^2 quadrats spaced at 10 m intervals along the 50 m transect (Neldner et al., 2019). In plots located on saltmarshes bare ground was included as a pseudospecies. The non-detailed plots were used to validate mapping consequent to this classification.

There were 70 species included in the analysis. Some taxa that were not consistently identified to species were amalgamated to genus level. Plant nomenclature follows (Bostock & Holland, 2018).

Plant community classification analysis

Plant communities were identified following the standardized quantitative classification methods adopted by the Queensland government and outlined by Addicott, Newton, et al., (2018) and Neldner, Wilson, et al., (2019). This was a four-step process involving; 1) excluding species with <1% contribution to total foliage cover in any plot (Addicott, Newton, et al., 2018), resulting in 42 species in the analysis; 2) recognizing differences in structural formation by multiplying %PFC of each species by vegetation layer height to form species importance values for each plot (Addicott, Laurance, et al., 2018); 3) forming clusters with these species importance values from a similarity matrix (Bray-Curtis dissimilarity coefficient on square-root transformed data) then agglomerative hierarchical clustering with unweighted pair group mean average; and 4) determining the cluster division level to form plant communities using a combination of three evaluators: a) SIMPROF (K. R. Clarke et al., 2008), b) Indicator Species Analysis (Dufrêne & Legendre, 1997) in the 'labdsv' R package (Roberts, 2013); and c) the ability of the classification to predict foliage cover distribution. This last method used generalised linear models in a multivariate framework (Lyons, 2017; Lyons et al., 2016). All analyses were done using PRIMER-E software version 7.0.13 (K. R. Clarke & Gorley, 2015) (Clarke & Gorley, 2015) and in the R environment (R Development Core Team, 2014).

The final plant communities were incorporated into the Queensland government classification system by a peer-review committee, and in to regional ecosystem mapping (RE mapping) for Queensland which is a state-wide mapping dataset (available from http://qldspatial.information.qld.gov.au/catalogue/custom/index.page). This mapping had a minimum mapped area of 5 ha and a line accuracy of 100 m (Neldner, Wilson, et al., 2019) and was used to estimate the extent of mangrove forest and saltmarsh communities in the bioregion.

Estimating the carbon storage and sequestration capacity of the intertidal communities

Intertidal communities are recognized as areas of high C storage and sequestration capacity and the potential of these communities in CYP was estimated using the mapped areas of the identified plant communities and figures from the literature. For C storage the figures for above and below ground calculated by Donato et al., (2011) and Macreadie et al., (2017) were used. Carbon storage of mangrove forests of the Indo-West Pacific region estimated by Donato et al.,(2011) are applicable to this study area which lies in the same region, containing estuarine and oceanic mangrove forests (Duke, 2006) of the same species and structure. The mangrove forests in this study had average height ranges from 6.5 - 14 m, stem diameter ranges from 3.6 - 18 cm and canopy closure ranges from 27% to 80% (Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP). When compared with those used by Donato et al., (2011), they fell within the categories of small to intermediate stature (tree diameter <10cm - 20cm or canopy height <4m - 15m). The same soils depths for 'estuarine / deltaic' (~3 m) and 'oceanic / fringing' (~1.2 – 1.7m) were also assumed. As they found significant difference between carbon stores in 'estuarine / deltaic' forests and 'oceanic / fringing' forests, the carbon storage estimates for forests in this study were calculated as: 'estuarine /deltaic' forests (above ground 63.4 Mg C/ha *(± 18.1 s.e.m.),* below ground 925.3 *(± 269.6 s.e.m.)(n = 6)*); 'oceanic / fringing' forests (above ground 122.2 Mg C/ha *(± 15.0 s.e.m.),* below ground 618.4 *(± 50.7 s.e.m.) (n = 8)*) ; all mangrove forests calculated using an all sample mean (above ground 97.0 *(± 21.0 s.e.m.),* below ground 722.7 *(± 174.3 s.e.m.),* total 819.7 *(± 177.3 s.e.m.) (n = 14)*) (Donato et al., 2011, Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP). For estimating the carbon storage of the saltmarshes, the Australian average estimate of 77.92 *(± 3.35 st. dev.)* Mg C / ha at 30cm depth (Macreadie et al., 2017) was used. To calculate C sequestration capacity estimates, the national figure for saltmarshes given by Macreadie et al, (2017) was used

(0.55 ±*0.02* Mg C / ha / yr). For mangrove forests two figures were available, providing a range. A figure from mangrove forests in sub-tropical south-east Queensland with similar communities provided the lower estimate (0.76 ±*0.02* Mg / ha / yr) (Lovelock et al., 2014) and a figure from tropical north Queensland with similar communities, provided an upper estimate (1.8 Mg / ha /yr) (Brunskill et al., 2002). These figures were extrapolated across the mapped area of mangrove forests and saltmarshes to provide estimates of C sequestration capacity across CYP.

The C store estimates were extrapolated across the RE mapping differently for saltmarshes and mangrove forests. For saltmarshes, the areas of the three saltmarsh communities were combined. Mangrove forests, however, needed an additional classification to distinguish between ‘oceanic / fringing’ and ‘estuarine / deltaic’, hence the following protocol was developed. Estuaries were defined as zones which "permanently or periodically open to the sea and receive[ing] at least periodic discharge from a river" (Potter, Chuwen, Hoeksema, & Elliott, 2010) and which are separated from the ocean (Kench, 1999). Each of Donato et al., (2011) sites were inspected on Google Earth to understand the context and geomorphological position of their ‘estuarine / deltaic’ and ‘oceanic / fringing’ mangrove forests for comparison to those mapped in the RE mapping. The following set of guidelines was then developed to map areas of ‘estuarine / deltaic’ and ‘oceanic / fringing’ mangrove forests in CYP:

1. Where there was a barrier (either bedrock, wave-built sand dunes or alluvial floodplain) between mangrove forests in an estuary and those on a coast, coastal mangroves were classified as ‘oceanic / fringing’ (Kench, 1999, Appendix 6.2 Examples of implementation of guidelines for determining estuarine mangroves and oceanic mangroves using the regional ecosystem mapping of Queensland).
2. Mangrove forests with no obvious freshwater source (at a scale of 1:5,000 on Earth-i 2017, 80 cm resolution satellite imagery) were classified as ‘oceanic / fringing’.
3. Mangrove forests with reef directly offshore were classified as ‘oceanic / fringing’.

4. On some Torres Strait islands, mangrove forests from an estuary reached to the shore-front with no barrier AND there was a fringing reef. In this situation the boundary of the fringing mangrove community defined ‘oceanic / fringing’ forest. Inland from this was classed as ‘estuarine / deltaic’ forest (Appendix 6.2).
5. Because Kench (1999) specifically described mangrove forests in the Great Barrier Reef lagoon as pro-grading deltas these shoreline forests were classed as ‘estuarine / deltaic’.
6. Donato et al., (2011) found oceanic soils had higher C concentrations than estuarine soils in the mangrove forests of similar stature to our study area. Therefore, a conservative approach was adopted and where there was doubt, mangrove forests were classed as ‘estuarine / fringing’.

To contextualise the C storage of mangrove forests in CYP, total C storage for the most widespread woodlands and the rainforests in the bioregion was calculated. The most widespread woodlands are the *Eucalyptus tetrodonta* woodlands (Accad, Neldner, et al., 2019) and combined total C storage (above and below ground) for these was estimated using research for the *E. tetrodonta* ± *E. miniata* woodlands in the nearby Northern Territory; 75 Mg C / ha *(± 40.2 st.dev.)* (Collins et al., 2009). Rainforests, which are equivalent to IPCC ‘tropical wet’ climate zone, cover ~500 000 ha in CYP. There were two estimates of stored C / ha available for rainforests, 213 Mg C / ha (IPCC, 2014) and 231 Mg C / ha *(± 75 st.dev.)* (H. Keith, Mackey, & Lindenmayer, 2009) which were used as a lower and upper limits.

CO_2 equivalents are widely used as a standard for comparing measures of carbon from varying sources (Commonwealth of Australia & Department of the Environment and Energy, 2018). To standardize the figures of C content to CO_2 equivalents the C content was multiplied by 3.67, the ratio of C to CO_2 based on molecular weight (Pendleton et al., 2012).

Assessing the vulnerability of mangrove forests to El Nino-driven dieback

The mangrove dieback reported by Duke et al., (2017) occurred along a low-relief coastline with adjacent extensive saltmarsh plains and a dry to semi-arid climate. The potential dieback vulnerability of mangrove forests in CYP was therefore estimated by confining calculations to two areas with similar geomorphological settings and climatic ranges (Danaher, 1995; Neldner, 1999). These were the south-western CYP coastline reaching from Weipa to the bioregional boundary, and the coastline of Princess Charlotte Bay in the east (Figure 6.1 Cape York Peninsula bioregion, north-eastern Australia. The Gulf of Carpentaria is on the western side and the Great Barrier Reef fringes the east. The semi-arid Gulf Plains bioregion lies to the south and west, while the wet-humid bioregion of the Wet Tropics is to the south and east.. The similarity in climate of these areas to those of dieback in the Gulf of Carpentaria was compared using the modelled variables of precipitation of the wettest and driest quarter and seasonal variability (Coefficient of Variation), and the average temperature of the warmest and coldest quarter and seasonal variability (Xu & Hutchinson, 2013); all variables known to influence mangrove distribution and growth (Duke, 2006).

To estimate an area of mangrove forest potentially vulnerable to dieback along the south-west and Princess Charlotte Bay coastlines the patterns of mangrove dieback found by Duke et al., (2017) were modelled using the RE mapping and GIS analysis. Duke et al., (2017) found five distinct patterns in the mangrove forest dieback; 1) an almost complete loss of forests on the oceanic shoreline, 2) dieback on the margins of mangrove forest where they bounded saltmarshes, 3) dieback on ecotones between landward margins of the mangrove forest communities, 4) dieback patches were 100 m - 200 m wide, and 5) forests along estuarine water courses were less affected. Four of these patterns were able to be modelled to identify areas of mangrove forest fitting the patterns. To apply pattern 1, the total area of ‘oceanic / fringing’ forest in the coastal plains was included as equivalent to oceanic shoreline forests (252 ha). To apply patterns 2, 4 and 5, the boundary between the mangrove forests and saltmarsh was buffered by 100m (the minimum width of dieback found

by Duke et al., (2017) and then categorized in to 1) oceanic-shoreline; 2) landward margins of mangroves abutting saltmarsh; or 3) estuarine-watercourses. Buffering along the landward boundaries between forest communities (pattern 3) was not possible as the RE mapping was not detailed enough to show boundaries between individual communities and many areas were attributed with multiple communities.

To test this conceptual model for estimating the vulnerability of mangrove forests to dieback, random points in the mapped areas of each category were produced and investigated for the presence of dieback. The ratio of area in each category was used to determine the number of points to be investigated in each category (oceanic-shoreline = 2.9%, estuarine-watercourses = 71.9% and landward margins = 25.3% of total area) with 150 m specified between points and a maximum of five points per polygon. However, we chose 15 points in the oceanic-shoreline category to achieve statistical robustness, as the proposed method produced only five (Table 6.3). These points were then buffered by 100 m (the accuracy limit of the mapping linework) and visually inspected for evidence of dieback using Earth_i_80 2016 and 2017 satellite imagery at a scale of 1:2,500. This imagery shows mangrove dieback resulting from the 2015-16 El Nino event. Areas of doubt were further inspected on historical air photos using 'QImagery' (https://qimagery.information.qld.gov.au/) and historical satellite imagery on Google Earth to look for differences in patterns.

The findings of this investigation were used to estimate the area of dieback that occurred on these coastlines during the 2015-16 El Nino event. Differences in the amount of dieback between the south-western coastline and Princess Charlotte Bay area, and between different categories were tested using Fisher's exact test (GraphPad Software, 2017). To calculate an area of dieback for each category the percentage of points showing dieback was extrapolated to the percentage of the area of each category. However, Duke et al., (2017) found almost total loss of coastal shore-line mangrove forests. Consequently, where dieback was found in these forests the total area was used. As a control test for this conceptual model, results were compared to findings by Accad, Li, Dowling, Neldner, &

Turpin (2019) who assessed dieback from the 2015-16 El Nino event across all the mangrove forests along the western coastline of CYP bioregion using manual assessment of satellite imagery.

Updating species richness in Cape York Peninsula bioregion

The floristic diversity and species richness of the mangrove communities in north-eastern Australia have been studied in great detail (Duke, 1992, 1998, 2006) but not described in the bioregional context commonly used in Australia (Thackway & Cresswell, 1995). The mangrove species richness was updated to this context using previously published information (Duke, 1992; Duke & Kudo, 2018) and specimen collection records from Queensland and Cairns herbaria (the two herbaria with up-to-date curation of plant taxa in the bioregions) (Australian Virtual Herbarium & HerbRecs databases accessed 19th October 2018). Only 'true mangrove' species (Tomlinson, 2016) were considered. The species richness of saltmarshes in CYP was updated by comparing recent herbaria records to the list of saltmarsh species in bioregions provided by Saintilan (2009b).

Results

Plant Communities

The analysis of intertidal plant communities split the mangrove forests from the saltmarshes (Figure 6.2), with five mangrove forest and three saltmarsh communities recognized. (Figure 6.2, Appendix 6.3 Descriptions of the mangrove forest and saltmarsh communities of CYP bioregion).

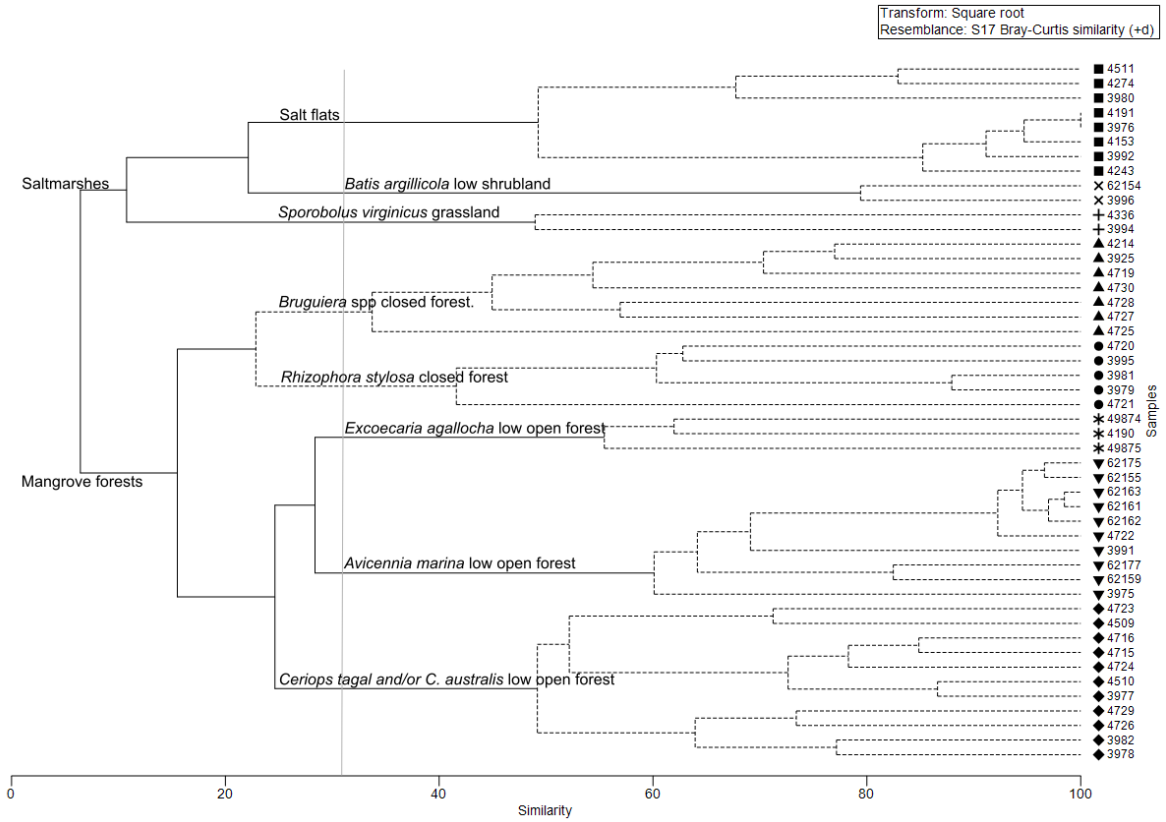

Figure 6.2 Dendrogram of inter-tidal plant communities on CYP. Solid lines show groups that were significantly different to each other using SIMPROF evaluator (K. R. Clarke et al., 2008). The grey vertical line shows the final clusters accepted as communities using Indicator Species Analysis (Dufrêne & Legendre, 1997). Clusters are labelled with the dominant species of the community.

Saltmarshes

The saltmarshes covered an estimated 1 054 km^2 (Table 6.1) making up ~8% of Australia's saltmarsh area (Macreadie et al., 2017) and ~15% of saltmarsh area of Queensland (Accad, Neldner, et al., 2019). This is the third largest area of saltmarsh in Queensland after the Gulf Plains bioregion to the west and the Brigalow Belt bioregion in the south of the state. The communities recognized were; saltflats with patches of herbland (921 km^2), *Sporobolus virginicus* grassland (133 km^2) and *Batis argillicola* shrubland (<1 km^2) (Appendix 6.3 Descriptions of the mangrove forest and saltmarsh communities of CYP bioregion). There were no new records of saltmarsh species for the bioregion.

Mangrove Forests

The mangrove forests (Figure 6.2 Dendrogram of inter-tidal plant communities on CYP. Solid lines show groups that were significantly different to each other using SIMPROF evaluator (K. R. Clarke et al., 2008). The grey vertical line shows the final clusters accepted as communities using Indicator Species Analysis (Dufrêne & Legendre, 1997). Clusters are labelled with the dominant species of the community.Appendix 6.3 Descriptions of the mangrove forest and saltmarsh communities of CYP bioregion) covered an estimated area of 1 539 km^2, which was ~32% of mangrove forests in Queensland and the most extensive of all bioregions in the state (Accad, Neldner, et al., 2019). National and global figures for the area of mangrove forests vary. Nationally, the most recent estimate is ~11 142 km^2 in 2017 (Lymburner et al., 2020). Globally, the most recent estimates range from between ~70 096 km^2 in 2012 (Hamilton & Friess, 2018) and ~137 760 km^2 in 2000 (Giri et al., 2011). CYP therefore contained ~14% of the remaining mangrove area nationally and between ~1.1 - 2.2% globally.

The most extensive communities were those dominated by *Rhizophora stylosa* and / or *Bruguiera* species (Table 6.1), occupying the low-to-mid tidal position at the downstream end of estuaries (Duke, 2006). The survey plots indicated *Bruguiera* species occur in small areas at the landward edges of the *Rhizophora stylosa* zone. These species are extremely difficult to distinguish from remote-sensed imagery and are therefore combined for the RE mapping. The next most extensive community was *Ceriops tagal* dominated, occupying the mid-to-high tidal position in the middle section of estuaries (Duke, 2006), with the less extensive *Avicennia marina* dominated community mainly occupying the high tidal position in the same section of estuaries (Duke, 2006). The least extensive community was *Excoecaria agallocha* dominated, occurring in the mid to upper reaches of estuaries and rivers straddling the full tidal range (Duke, 2006). The last three communities are maintained as separate communities in the RE mapping. There were 32 species of 'true' mangrove recorded in the bioregion (Appendix 6.4. Species list of mangroves in Cape York Peninsula bioregion)

representing 82% of mangrove species diversity in Australia and 48% of the global mangrove species diversity.

Table 6.1 Estimates of extent of mangrove forest and saltmarsh communities in CYP bioregion. Mangroves have been divided into 'estuarine / deltaic' and 'oceanic / fringing' based on RE mapping and guidelines outlined in the text.

	Total (ha)	*% of total mangrove forest area*	Estuarine (ha)	*% of community area*	Oceanic (ha)	*% of community area*
Mangrove forests						
Rhizophora stylosa or *Bruguiera* spp. closed forest	83 989	*54.6*	64 541	*76.8*	19 448	*23.2*
Avicennia marina +/- *Ceriops tagal* low open forest	25 980	*16.9*	22 637	*87.1*	3 343	*12.9*
Ceriops tagal and / or *Ceriops australis* low open forest	41 486	*27.0*	40 933	*98.7*	553	*1.3*
Excoecaria agallocha +/- *Aegiceras corniculatum* low open forest.	2 476	*1.6*	2 414	*97.5*	62	*2.5*

Total	153 931		130 525	23 406

Saltmarsh		*% of total saltmarsh area*
Sporobolus virginicus closed tussock grassland	13 252	*12.6*
Sparse herbland or bare saltpans on salt plains and saline flats.	92 086	*87.4*
Batis argillicola low shrubland	6	*0.01*
Total	105 354	

Carbon storage and sequestration capacity estimates of mangrove and salt marsh communities

Using the all sample mean, the mangrove forests were estimated to store a total of 126.2 *(± 27.3 s.e.m.)* Tg of C, with 14.9 *(± 3.2 s.e.m.)* Tg C above ground and 111.3 *(± 26.8 s.e.m.)* Tg C below ground (Figure 6.3 Estimated C stores in the inter-tidal communities of CYP. Forests are mangrove forests. 'All forests' was calculated using a sample wide mean. Estuarine forests and oceanic forests were mapped according to guidelines outlined in the text.Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP). This compared to the most widespread woodlands of CYP (*Eucalyptus*

tetrodonta woodlands), occupying 16 times the area and storing ~1.5 times the C, and to the rainforests covering 3.4 times the area and storing ~0.89 - 0.97 times the C (Table 6.2). The saltmarshes stored an estimated 8.3 *(± 0.4 st. dev.)* Tg of C (Figure 6.3 Estimated C stores in the inter-tidal communities of CYP. Forests are mangrove forests. 'All forests' was calculated using a sample wide mean. Estuarine forests and oceanic forests were mapped according to guidelines outlined in the text., Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP). Thus, the intertidal communities in CYP stored ~ 134.5 Tg C, an estimated equivalent ~493.47 Tg of CO_2. The C sequestration capacity of the mangrove forests was estimated to be between 0.12 (*±0.02 st dev*) – 0.28 Tg C / ha/ yr and of the saltmarshes 0.06 (*±0.02 st dev*) Tg C /ha /yr. The intertidal communities of CYP therefore sequester ~ 0.18 – 0.34 Tg C / ha / yr.

Table 6.2 Area and estimated total carbon storage of mangrove forests compared to other forests on CYP. The range for C stored in rainforest is derived from H. Keith et al., (2009) and IPCC (2014) default figure for the 'tropical wet' zone.

Forest type	Area (ha) *% of CYP area*	Multiple of area of mangrove forests	Total C (Gg) *(error)*	Multiple of C stored in mangrove forests
Mangrove forests	153 931 *1.2%*		126 177 *(+/- 27,286.3 s.e.m.)*	
E. tetrodonta woodlands	2 468 946 *20.2%*	16	185 500 *(+/- 99,428.0 st.dev.)*	1.5
Rainforest range (Keith 2000, IPCC (2014) default)	528 343 *4.3%*	3.4	112 537 – 122 047 *(+/- 39,625.8 st.dev.)*	0.89–0.97

Most mangrove forests in CYP were 'estuarine / deltaic' (1 305 km^2), with 'oceanic / fringing' mangroves (234 km^2) confined to island perimeters and the south-western coastline (Table 6.1). 'Estuarine / deltaic' mangroves stored an estimated 120.7 *(± 35.2 s.e.m.)* Tg of C, with 8.3 *(± 2.4 s.e.m.)* Tg above ground and 112.5 *(± 33.9 s.e.m.)* below ground. 'Oceanic / fringing' mangroves stored 17.3 *(± 1.2 s.e.m.)* Tg of C, with 2.8 *(± 0.30 s.e.m.)* above ground and 14.5 *(± 1.2 s.e.m.)* below ground (Figure 6.3, Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP). As the distribution of 'estuarine / deltaic' and 'oceanic / fringing' mangroves are not possible to display at a meaningful scale the zipped GIS shapefile of the mapping is provided for download (Appenidix 6.5).

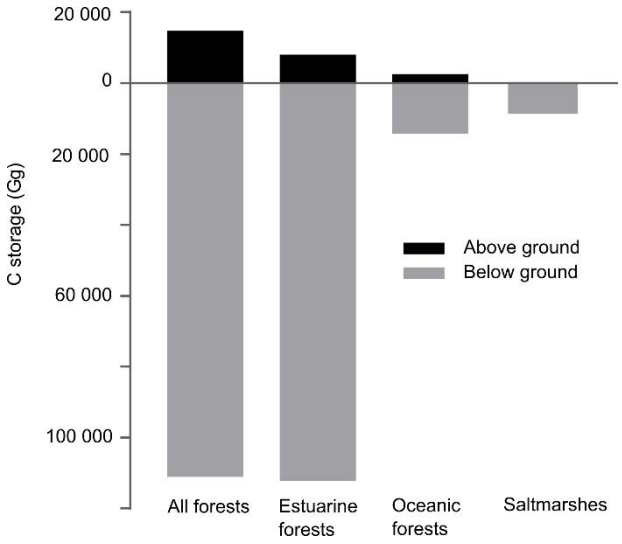

Figure 6.3 Estimated C stores in the inter-tidal communities of CYP. Forests are mangrove forests. 'All forests' was calculated using a sample wide mean. Estuarine forests and oceanic forests were mapped according to guidelines outlined in the text.

Assessing the vulnerability of mangrove forests to El Nino driven dieback

The conceptual model for identifying areas of mangrove forest as potentially vulnerable to dieback from events such as the 2015-16 El Nino was supported with dieback found at 4% of the 661 random points in the buffer areas. An area of 6 558ha in total was identified as being potentially vulnerable to dieback, with 3 647 ha along the south-west coastline and 2 911 ha in the Princess Charlotte Bay area. Dieback did not occur evenly across all

categories or between the environmentally similar east and west coast (Table 6.3). There was no dieback recorded in the Princess Charlotte Bay area (east coast), whereas 6% of investigated locations recorded dieback on the south-western coastline ($p < 0.001$). In this area, there was no mangrove dieback recorded in the estuarine watercourses, but dieback was detected in 100% of all points in the oceanic-shoreline forests and 13% of points along the landward margin between mangrove forests and saltmarsh (Table 6.3). This supports the findings of Duke et al., (2017) that there was less dieback in the estuarine watercourses than in the other categories. The frequency of dieback in the oceanic-shoreline forests was significantly greater than in those in the landward margin ($p < 0.0001$). Extrapolating these percentages of point death to the area within the buffer zone results in a potential 110 ha of landward margin forests suffering dieback in the 2015-16 El Nino event (pattern 2). Including the total area of oceanic-shoreline mangrove forests, which was 252 ha (pattern 1), resulted in an estimated total of 360 ha suffering dieback. This compares with the 479 ha of dieback found by Accad, Li et al., (2019). However they also found dieback along the western coastline of CYP north of the area of investigation in this study.

The mangrove forest communities most vulnerable to dieback appeared to be *Avicennia marina* forest (Figure 6.4) as the majority of the oceanic-shoreline forests consisted of this community. This is supported by Accad, Li et al., (2019) who found this was the primary mangrove forest which suffered dieback. The other two communities mapped in the buffer areas, and therefore also potentially vulnerable to dieback, were *Rhizophora stylosa* and / or *Bruguiera* low forests and *Ceriops tagal* low forests.

Table 6.3 Positive dieback in buffer zones of mangrove forests in CYP. Random points were generated in each buffer zone category and an area of 100m radius inspected for signs of dieback at each point.

	Princess Charlotte Bay area of CYP			south-western coastline of CYP		
Buffer zone category	Area in buffer (ha)	Number points	Number of points with *+ve* dieback	Area in buffer (ha)	Number points	Number of points with *+ve* dieback
Oceanic-shoreline	149	15	0	39	15	15
Landward margin	827	47	0	829	114	15
Fringing estuarine	1935	176	0	2779	294	0
Total	2911	238	0	3647	423	30

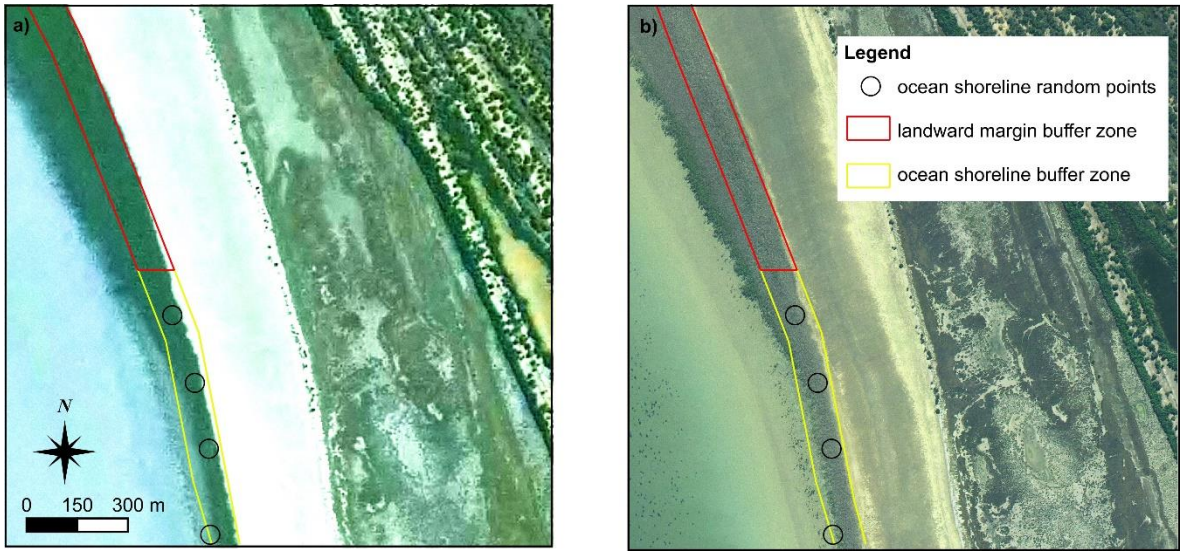

Figure 6.4 Satellite image showing dieback before and after the 2015-16 El Nino event (the green or light grey zone between the sand on the right and ocean on the left). **a)** = before. The dark green indicates live mangroves (SPOT imagery 2012, 2.5m resolution). **b)** = after. The grey indicates dead mangrove and can be clearly seen (Earth_i imagery, 80cm resolution). A 100m width buffer zone was applied along the mangrove forest / saltmarsh boundary and categorised as either landward margin, ocean-shoreline or estuarine-watercourses. Random points in mapped polygons of each category were visually checked for sign of dieback.

Discussion

Along the 7 480 km coastline of Cape York Peninsula in north-eastern Australia, approximately as long as that of Italy, the intertidal communities were classified to identify their floristic assemblages, potential C storage and sequestration capacity and vulnerability to extreme climate events. Three saltmarsh communities and five mangrove forest communities were identified, with the mangrove forests containing 82% of the Australian and 48% of the global mangrove diversity. These mangrove forests covered ~1 539 km^2 accounting for ~14% of all of mangrove forests in Australia and 1.1 - 2.2% mangrove forests globally. The saltmarsh communities covered approximately 1 054 km^2, accounting for ~8% of the saltmarsh area of Australia. Approximately 134.5 Tg C was estimated to be stored, and between ~0.18 – 0.34 Tg C to be sequestered per year in the intertidal communities, with ~360 ha of forest as potentially vulnerable to dieback. This study provides baseline

information on the mangrove and saltmarsh communities in the bioregion, important for conservation management and re-dressing some of the data deficiencies of these ecosystems identified by Himes-Cornell et al., (2018); Rogers et al. (2018) and others.

The mangrove forest communities recognized in this study are similar to others recognized across the Indo-West Pacific region, highlighting their diversity at a global level. They are dominated by the same genera as in areas such as the Andaman - Nicobar Islands in India (*Rhizophora, Bruguiera, Ceriops, Avicennia*) , Darwin Harbour in Australia (*Rhizophora*, *Ceriops*) , the Sundarbans in Bangladesh (*Ceriops, Excoecaria*) , Micronesia (*Rhizophora, Sonneratia, Bruguiera*) , Sarawak (*Rhizophora, Ceriops, Xylocarpus*) , Indonesia (*Aegiceras, Rhizophora, Avicennia*) and Papua New Guinea (*Rhizophora, Bruguiera, Avicennia, Ceriops, Sonneratia)* (Ashton & Macintosh, 2002; Aslan et al., 2016; Brocklehurst & Edmeades, 1996; Hinrichs, Nordhaus, & Geist, 2009; Kauffman, Heider, Cole, Dwire, & Donato, 2011; Ragavan et al., 2015; Rahman, Khan, Hoque, & Ahmed, 2015). One notable difference is the presence of *Nypa fruticans* (mangrove palm) dominated forests immediately to the north in Papua New Guinea and Sarawak (Ashton & Macintosh, 2002; Aslan et al., 2016; Robertson, Daniel, & Dixon, 1991) and their absence in this study. *Nypa fruticans* does occur in CYP (Appendix 6.4) but in small scattered patches below the scale of this study. A direct comparison between the number of mangrove forest communities in different regions is difficult as communities in each study have been classified using different approaches and attributes (for example Hickey et al., (2018); Kauffman et al.. (2011); Owers et al., (2018); Ragavan et al., (2015) (2015); Sarker et al., (2019)), and many areas of the world's mangrove forests are heavily disturbed while those in this study area are intact (<1% lost between 1960 and 2017 (Accad, Neldner, et al., 2019)). Despite this, a community that is not recognized here, but is in other regions (Aslan et al., 2016) and in an earlier study in CYP (Danaher, 1995) is that of a mixed-species mangrove forest which appears to be in transition zones. This may indicate a gap in the survey coverage in this study, although the classification system used by the Queensland government does not recognize transition

zones as separate communities. Community diversity of mangroves forests and saltmarshes decreases and increases respectively with latitude (Duke, 2006; Saintilan, 2009a). It is therefore not surprising that this study recognized only three saltmarsh communities and there are only 21 saltmarsh species listed for CYP (Saintilan, 2009a) in comparison with south-eastern Australia which has seven communities (Boon et al., 2015) and 63 species (Saintilan, 2009a).

This study provides the first estimates of C store and C sequestration capacity of the intertidal communities at a regional scale in north-eastern Australia. It also presents the first spatial delineation of 'estuarine / deltaic' and 'oceanic / fringing' mangroves in the Indo-West Pacific region, a step towards filling a data deficiency identified by Donato et al., (2011). The estimate of ~134.5 Tg C stored in the intertidal communities is likely to be conservative for three reasons. Firstly, there is no published typology for categorizing mangrove forests in to 'estuarine / deltaic' and 'oceanic / fringing' and the conservative approach adopted to categorizing forests as 'oceanic / fringing' in this study means a possible underestimation in the area of these forests. A standardized typology would help improve this categorization. Secondly, while C storage figures were calculated for mangrove forests based on the geomorphological characteristics of oceanic and estuarine position, as recommended by Donato et al., (2011), other studies have found differential rates of above and below ground carbon stored between mangrove species (Atwood et al., 2017; Hamilton & Friess, 2018; Laffoley & Grimsditch, 2009). These can be due to differences in root structure (Pérez, Libardoni, & Sanders, 2018), height and girth of different species (Hickey et al., 2018). Thirdly, the soil C estimates for both the mangrove forests and the saltmarshes have been estimated from the best available information and highlights a lack of plot data in the region. The lack of plot data in northern Australia also affects the estimates of C sequestration capacity. In the case of the saltmarshes the estimate for CYP was calculated using the national average, which in itself was calculated using data collected in different climatic regions (Macreadie et al., 2017). There was no data available for tropical regions (Serrano et

al., 2019). The estimate for mangrove forests was similarly based on the best available information with data from one plot available in northern tropical Queensland and one in southern sub-tropical Queensland. Until a sufficient sample size of plot data is available for Australian tropical intertidal systems land-use planning will rely on best estimates such as this study. This data deficiency (Kauffman et al., 2020; Macreadie et al., 2017; Serrano et al., 2019) points to an important direction for future work. The collection of detailed data quantifying the C stored above and below ground, and the C sequestration rates in the intertidal communities in the tropical areas of Australia will improve C accounting and aid in developing appropriate climate change mitigation strategies (Kelleway et al., 2017).

Despite these limitations in accurately estimating the C store and sequestration capacity in CYP this study highlights the importance of these intertidal communities in the context of the C value and emissions of Australia. Their estimated C store would be valued at $6.8 billion on the Australian carbon market as at December 2018 (http://www.cleanenergyregulator.gov.au/ERF/Auctions-results/december-2018 , accessed 22/4/19), and the C sequestered annually by these communities would be valued at between AU$8.7 - $17 million. The C sequestered would be equivalent to between ~0.03% - 0.06% of the national C emissions of Australia in the financial year to June 2018 (533.7 Tg CO_2), while the C store would be equivalent to ~92% of emissions in that year (Commonwealth of Australia & Department of the Environment and Energy, 2018). The potential C store is greater than the 'tropical wet' IPCC category forests (rainforests) of the bioregion, which cover 3.4 times the area, and two-thirds the estimate potentially stored in the widespread *Eucalyptus tetrodonta* woodlands of the bioregion, which cover 16 times the area. Future work to determine allometric equations for the dominant mangrove species, specific soil C storage capabilities for all the intertidal communities in 'estuarine / deltaic' and 'oceanic / fringing' situations and finer scale mapping using our classification will refine these C store estimates for the bioregion.

The identification of areas at potential risk of dieback using mapping and GIS analysis provides a viable alternative to other common methods such as satellite imagery classification (Duke et al., 2017) or manual interpretation of remotely sensed imagery (Accad, Li, et al., 2019). In this study, only areas of comparable geomorphology were included, however Accad, Li et al (2019) recorded dieback in catchments to the north of this area. This may indicate the areas of geomorphological and climatic similarity to those in Duke et al. (2017) are not fully captured by Danaher (1995) or Neldner (1999) or it may indicate that dieback occurred in differing geomorphological and climatic situations. Finer scale mapping and inclusion of more catchments are needed to refine the model.

The estimate of ~360 ha of dieback across the 24 188 ha of mangrove forest in the two areas with similar geomorphological and climatic situations to those reported by Duke et al. (2017) equates to a 1.5% areal loss, much lower than the 6% reported for the adjacent Gulf of Carpentaria. However, of the two locations, Princess Charlotte Bay and the south-western coastline of the bioregion, evidence of dieback was found only on the south-western coastline. The estimate of ~360 ha of potential dieback equates to a 2.5% loss of the 14 371 ha of mangrove forest in this area which forms the south-east coastline of the Gulf. This is much lower than the 9% reported for the western coastline, but similar to the 3% reported for the southern coastline of the Gulf (Duke et al., 2017). A comparison of environmental conditions between the locations suggests that the south-western coastline of CYP suffered the same prolonged drought conditions, extreme temperatures equivalent to ‘highest on record’ and drop in sea level as the study region of Duke et al. (2017). Princess Charlotte Bay, however differed in not having the same extreme temperatures or drop in sea level but did experience the same severe drought conditions.

The estimated ~360 ha dieback in this study is likely to be an underestimate as not all the dieback patterns found by Duke et al. (2017) were able to be incorporated into the GIS spatial model. This potential underestimate was supported during the visual inspection of imagery where there appeared to be areas of dieback along the landward edges between

communities, the pattern not incorporated. Additionally, only the lower limit of the width of the dieback zones found by Duke et al. (2017) was used. They found 100 – 200 m wide dieback zones, while 100m was used as the buffer limit in this study. Notwithstanding this, if these areas did not recover there is potentially between 0.001 – 0.002 Tg C / yr not sequestered and if the estimated C stored were to be mobilized, there is a possible 1.05 Tg CO_2 eventually released back to the atmosphere. It is possible that in more extreme El Nino related climate events larger areas of the 6 558 ha of vulnerable mangrove forests may experience dieback.

While no mangrove dieback was recorded in Princess Charlotte Bay during the 2015-16 El Nino event, there are reports of dieback in the supratidal communities adjacent to the saltmarshes in this area. These communities are not mangroves but dominated by *Melaleuca* species, grasslands and sedgelands. Evidence of dieback of these communities, such as dead tree stumps and changes observed on aerial photographs, suggest dieback has been occurring since at least 1955. Because of lack of data on sea levels, tidal surges and other environmental variables along this remote coastline understanding the contribution of climate change to this dieback is problematic. Preliminary estimates have found ~400 ha of dieback, with saltmarsh encroaching into the supra-tidal communities and changing the boundary between intertidal flats and coastal plains (S. Thompson, pers. com., April 2019). Based on this information the conservation status of the *Melaleuca saligna* dominated community was changed to Endangered at a State level (https://apps.des.qld.gov.au/regional-ecosystems/details/?re=3.3.13 accessed 16th February 2020). Delineating the extent of the changes in these coastal communities in the Princess Charlotte Bay area is the focus of ongoing work, with early work also showing a possible encroachment of mangrove forest communities inland into saltmarshes (S. Thompson, pers. com., April 2019). There are also other reports of the retreat of supra-tidal communities occurring in the Torres Strait Islands between Australia and Papua New Guinea (Stanton, Fell, & Gooding, 2009).

Coastlines are expected to be one of the most affected systems as a result of climate change (IPCC, 2014). One suggested impact is an increase in mangrove forest extent with a consequent increase in C sequestration (Kelleway et al., 2016), but this may be at the expense of supra-tidal communities with a possible loss of biodiversity. We suggest that to develop climate change mitigation strategies at a national and global level it is important that detailed documentation of changes to biodiversity and coastal communities in remote, intact areas is collected. This information will be useful for benchmark comparisons with other areas to understand the effects of climate change, and, the effectiveness of implemented mitigation strategies designed to counter the diffuse effects of climate change.

The identified data deficiencies in these systems in CYP are understandable given their extent, remoteness, sparse population, and data collection risks such as crocodile presence. Although this study is the first to give estimates of baseline C store, sequestration rates and dieback vulnerability, in order to maintain their global significance, the first priority is acquiring detailed baseline information. In areas that are safe, plot specific data to document above and below ground C stores and sequestration rates is urgently needed. This, complimented by more detailed mapping of the coastal ecosystems extent and change, will enable the second priority for their maintenance. This is to adopt management strategies at landscape and regional scales as it is unlikely that local solutions, such as dikes or intensive restoration projects, will be cost effective across such an extensive and remote area. Detailed baseline information will allow realistic quantification of the trajectory of ecosystem changes in biodiversity loss or C sequestration capacity as the boundaries between ecosystems change. For example, in the current Australian climate fire and water management are high priorities; both landscape scale management issues which are also increasingly important at a global scale. Understanding whether these systems are on a trajectory of change which includes changes in fuel moisture and potential increase in flammability, or, understanding whether proposals for agricultural or mining development with its accompanying extraction of fresh water from the system may exacerbate these

trajectory changes, can only come through having detailed baseline data. This information can also underpin landscape planning instruments such as incentivization schemes (for example Department of Environment and Science, (2020)), blue carbon accounting, planning regulations and landscape management strategies designed to enhance the resilience of these coastal systems.

Conclusions.

Degradation and destruction of saltmarshes and mangrove forests is a global issue (Murray et al., 2011). Our study highlights the global significance of the diversity, extent, C store and sequestration capacity of the intertidal systems of CYP and reinforces calls for the inclusion of coastal ecosystems in appropriate climate change mitigation strategies (Kauffman et al., 2020). Although these systems in the CYP region are not threatened by direct impact from the anthropogenic activities common elsewhere in the world, our results support previous work highlighting the potential vulnerability of intact intertidal systems to more diffuse threats, such as, for example, those from stronger El-Nino events (Duke et al., 2017; Feller, Friess, Krauss, & Lewis, 2017). These systems in CYP are currently largely intact, however CYP is seen as an area of high development potential (Department of State Development Infrastructure and Planning, 2014; Department of the Prime Minister and Cabinet, 2015) and the preliminary information in this study must be incorporated into planning instruments at State and National levels in order to maintain their global importance and enhance their resilience to projected climate changes (IPCC, 2014).

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Chapter 7 Synthesis and discussion

Many landscape and vegetation classification systems have been developed at local and regional scales, but a standardised system crossing administrative and geographical boundaries is widely recognised as most useful for broad-scale land management (De Cáceres et al., 2015; Jennings et al., 2009; Walker et al., 2013). The Queensland government, recognising this, uses the RE classification system and accompanying mapping as a state-wide standard landscape classification system. This system informs legislation at both the state and national levels, with decisions based on it having wide-ranging implications for the biodiversity of the state and people's livelihoods. As such it needs to be in line with global best practice. In many ways the RE system aligns with best practice (Chapter 1) however the current classification approach for identifying plant communities relies on expert-based (supervised) class definition procedures. To bring the RE system further in line with best practice a new classification approach based on un-supervised class definition procedures needed to be developed (De Cáceres et al., 2018).

I addressed this requirement by determining, testing and evaluating un-supervised class definition procedures consistent with the concepts and criteria of the RE classification system and combined them to develop a new classification approach. I demonstrated the robustness of my new classification approach by applying it in an ecological context to provide baseline conservation information. In this last chapter I synthesise my findings, make recommendations regarding implementation of the new approach and suggest future directions for research and continuing improvement to the RE system.

Final class definition procedures

The final suite of techniques, both un-supervised and supervised, making up class definition procedures for a new classification approach for the RE system have been synthesised from all chapters in my thesis, but particularly Chapters 3 and 4 (Table 7.1). The procedures

outline the different aspects of identifying communities to be considered and include considering the adequacy of the available dataset to sample the community diversity and the environmental variability; the un-supervised plot-grouping techniques; the internal evaluation techniques, and the external evaluation process. The specific process for the expert panel peer review assessment of new communities in the RE system needs to be formalised similarly to the USNVC revisions procedure (http://usnvc.org/revisions/ accessed 3/9/19). However, as part of the peer review assessment it is important that where the expert panel does not agree with the proposed communities, the panel's assumptions regarding the drivers for expected change in ecological function and patterns are explicitly identified (for example geomorphology or differences in ground layer composition). Ensuring these are then tested using quantitative evaluation techniques will help reduce the known biases of expert panels (Martin et al., 2012).

Table 7.1 Final suite of class-definition procedures making up a new classification approach for the RE classification system of Queensland.

Class Definition Procedure	**Procedures recommended from this study**	**Procedures adopted by Queensland Government**
Prior to class definition procedures		
Assess the adequacy of dataset	Test the dataset for adequacy in capturing: Community diversity Environmental variability **DOMAIN was used in this study, but other software programs may become available and recommended*	Accepted

Primary Vegetation Attributes		
Abundance measure	%cover	Accepted
Vegetation layers to include	Exclude ground layer from woody vegetation dominated plots	Accepted
Subset of species	Grasslands: exclude species contributing <8% to Total Foliage Cover Shrublands: exclude species contributing <1% to Total Foliage Cover Woodlands: exclude species contributing <10% to Total Foliage Cover	In all formations exclude species contributing <1% to Total Foliage Cover
Species weighting to incorporate structure	Multiply each species in a plot by vegetation layer height and sum across plot	Accepted
Plot-grouping technique		
Clustering algorithm	Option 1: Fuzzy noise clustering Option 2: Agglomerative hierarchical clustering (UPGMA linkage), with square-root transformation and Bray-Curtis coefficient	Option 2 accepted
Internal Evaluators	Geometric: SIMPROF Non-geometric: Indicator Species Analysis	Accepted

(to choose levels of cluster division forming communities)	Modelling: Optimus (Linear Regressoion using AIC)	
External evaluation		
(results to be presented for peer-review assessment)		
External evaluators	Landform	
	Geology	
	Geomorphology	
	Soils	Accepted
	Geographic distribution	
	Modelled datasets (for example radiometrics)	
Quantitative assessment	Test for floristic differences between plots using external evaluation categories and:	Accepted
	ANOSIM	
	Univariate statistical techniques	
Supervised assessment		
Compare	outcomes from other un-supervised techniques, for example:	
	Ordination	Accepted
	Cophenetic correlation	
Refer to	OPTIMUS (predicting distribution of species across dataset)	

	Existing classification systems	
	Non-detailed plot data and observational records	
Post peer-review		
Questions regarding proposed communities	Identify drivers of ecological function assumed to produce different floristic patterns	Accepted
Test differences in floristic composition using:	Univariate statistical techniques	
	Multivariate statistical techniques	Accepted
	GIS analyses	

Although this thesis outlines un-supervised classification techniques as a state-wide approach, the RE classification system will remain a semi-supervised system for two reasons. Firstly, the plant communities identified using the plot-grouping techniques are assessed through a peer-review process. This is common with all classification systems reviewed that included quantitative plot-grouping techniques (De Cáceres et al., 2018) and reflects the reality that a classification system must be interpretable by end-users. Secondly, there will always be communities in the RE system that have been identified using supervised techniques based on observational records, because of the sparse population and relatively low numbers of experts working in the vegetation science field in Queensland. As noted in Chapter 4, these communities represent ‘known unknowns’ and will be obvious targets for future survey work collecting detailed plots.

Whilst Gellie et al. (2018) argue for a move away from vegetation mapping as the basis of vegetation classification in Australia I would argue the opposite. In broad, uninhabited and

often inaccessible parts of the country where limited funding is available, vegetation mapping and classification systems developed in conjunction can provide adequate capturing of the environmental variables limiting plant species growth and community richness at a landscape scale (Chapter 2). Vegetation classification systems, based on vegetation mapping and a semi-supervised classification approach, will therefore continue to be an essential aid in land-management decision making.

Survey design

To understand and apply a classification system is it important to understand the efficacy of the data informing it and one of the common questions asked by users of the RE system is "How good is the sampling?" (H. Dillewaard personal communication April 21st, 2015.) The results of my work (Chapter 2) show that, as usual in the natural world, the answer is nuanced. Although the two-tiered preferential sampling design used by the Government captured the environmental variability and beta-diversity comprehensively, it does not adequately sample the species richness in the landscape. The RE system is therefore likely to identify most of the plant communities in a region but not adequately describe the floristic diversity of those communities. If the expectation is for the RE system to comprehensively capture the diversity of plant communities across the State, it succeeds. If the expectation is for the RE system to comprehensively describe the floristic diversity of communities, it is less than adequate. However, I would question the expectation that a landscape scale classification system be expected to fully describe the diversity of species whose distributions may be driven by factors operating at finer scales than those driving plant community patterns. As communities in the RE system are identified using the dominant species of the layers within a community, it is unlikely that sampling for the full floristic variability in the communities will lead to a better capture of the community richness in the landscape. Comprehensively sampling the floristic variability of communities will incur considerable costs in increasing the number of detailed plots to an adequate level. The benefit of these costs would have to be carefully considered.

Although many potential biases in the RE system are minimised by having standardised mapping and data collection methods (Neldner et al., 1995; Neldner, Wilson, et al., 2019) they still exist as the detailed plot data is collected at sites judged as representative of a plant community by the ecologist. Of the four widely recognised sampling designs for landscape scale vegetation survey, random-sampling, stratified random-sampling and systematic sampling are known to under-sample rare communities (Bell, 2013; De Cáceres et al., 2015) and to cover a smaller range of environmental extremes than preferential sampling (Roleček et al., 2007). My findings support this in that the environmental variability was comprehensively surveyed as was the beta-diversity on both landscapes tested using this preferential sampling design (Chapter 2). The primary drawback of a preferential sampling design is the loss of statistical power (Lájer, 2007) but, as is true of vegetation survey and mapping generally, one of the primary aims of the RE system is to distinguish and describe patterns rather than differences based on null hypothesis significance testing (De Cáceres et al., 2015). In these circumstances this loss of statistical power is acceptable.

The sampling design used by the Government within the RE system is two-tiered, with rapid qualitative data points (observational records) collected along all possible traverses in the landscape. These are used to locate the detailed data collection plots. I posit that this two-tiered approach is the reason that a small number of detailed plots (288 across ~54 000 km^2) comprehensively sampled the landscape variability and beta-diversity (Chapter 2). However, one potential drawback of this design is that collecting detailed vegetation plots is time-consuming (Neldner, Wilson, et al., 2019) and may be regarded as not cost effective. Mapping can be done using just observational records and supervised allocation of records to communities and if ecologists prioritise collecting observational data over detailed plots, then the adequacy of sampling within the RE system, and hence the quality of the mapping, will be reduced. To maintain the robustness of the RE system, and the mapping, it is therefore imperative that detailed vegetation plots are collected in any area perceived as representative of the surrounding plant community. Another problem common to collection of

both species and, most likely, community data is under-sampling of common species and communities (Garcillán & Ezcurra, 2011; Moerman & Estabrook, 2006); possibly a result of the human cognitive bias towards difference (Kahneman, 2011). This may lead to common communities in the landscape not being recognised by the un-supervised techniques in the class definition procedures. A possible check is to compare the sampling adequacy of the detailed plots with the observational records. This will ensure that an adequate number of detailed plots are collected and help reduce the tendency to collect observational records at the expense of detailed data plots.

Class definition procedures

The concepts and criteria of the RE system stipulate that communities are identified using the dominant and sub-dominant canopy species of the vegetation layer with the highest above ground biomass (Neldner, Wilson, et al., 2019). My work has shown that the concept of dominance differs between vegetation formations, with grasslands and woodlands requiring less sampling intensity than shrublands to capture the species useful for determining the plant community. It is widely accepted that concepts of dominance are not useful in species rich communities (Guo et al., 2018; Landucci et al., 2015). An extension of my work which would be useful in developing both a classification approach for species rich communities and a tool for defining survey effort, would be to quantify the level of species richness at which concepts of dominance break down.

Although my work showed dominance thresholds differ between formations the Government has decided to use the 1% threshold for dominance across all formations. The main implication of this is an increase in the ‘noise’ in the dataset identifying communities. One practical application of knowing appropriate levels of dominance is to provide a level of required sampling intensity to identify a community in a formation. Requiring all species contributing >1% to total foliage cover to be measured in all formations increases survey effort, without necessarily increasing the identification of ecological patterns (Pos et al.,

2014). This will increase the overall cost of the survey, or possibly, decrease the number of plots surveyed with a consequent reduction in the robust capture of community diversity.

Choosing a plot-grouping algorithm appropriate for a classification system is problematic, due in part to the large number of choices available. I put forward two suitable options: non-hierarchical fuzzy noise clustering (preferred); and agglomerative hierarchical clustering using Unweighted Pair Group Mean Averaging linkage, (adopted by the Government). One of the drawbacks of using agglomerative hierarchical clustering to identify communities is the difficulty of incorporating new plot data into the system without having to re-do the whole classification exercise (De Cáceres & Wiser, 2012), however new plot data will continue to be acquired as the RE mapping program is a continuing Government project. Re-running the whole classification exercise to incorporate new plot data is not ideal as the RE system needs to be stable from a social and political perspective. If this is not the case it has the potential to feed into the ongoing social debate about the validity of evidence-based decision making. Developing an approach for incorporating new information into the system in a way which maximises the stability of the existing plant communities, and identifies new ones, is therefore a priority. Various techniques exist (Oliver et al., 2012; Tichý et al., 2014) and my recommendation is to follow the approach adopted by administrations internationally rather than developing new techniques.

Outcomes of new approach

Communities identified using supervised techniques reflect the assumptions of ecologists regarding the drivers of ecological patterns. In my study area communities were previously identified using a supervised approach, and applying my new approach tested the assumptions underlying these. The differences between the outcomes from the new and old approach are highlighted in Chapter 4 and 5, with nearly half as many communities identified by my new approach; and these were more recognisable and useful for planning purposes. However, my new classification approach necessarily involves expert review of outcomes as part of the validation process, and this has the potential to re-introduce those assumptions

when experts do not agree with them. To help circumvent this, I recommend experts explicitly define their assumptions about drivers of different patterns (such as geomorphology, soils, moisture regimes etc) and to quantitatively test these (Chapter 4). Ensuring this testing is part of the expert review within the class-definition procedures will help reduce the subjectivity in applying expert assumptions. This will ensure that any community identified *a posteriori* to the plot-grouping techniques is evidence-based and scientifically defensible.

The strengths of using a quantitatively based classification approach to identify communities are multi-faceted. In Chapter 4, I demonstrate a potential use by outlining two ways in which the similarity and dissimilarity of communities can be shown. Having these quantified allows managers and end-users to understand the landscape and feed this information into downstream projects; for example, quantifying the conservation values of areas. Robust quantification of these similarities is not possible with communities identified using a supervised classification approach (Kent, 2012). In Queensland, another use of the RE system is in vegetation management legislation which allows modification of the landscape within the limits of structural variation of an RE (Queensland Government, 1999). The combination of, firstly, quantitatively identified communities and secondly, my work showing the adequacy of the Government's standard sampling design in capturing the beta-diversity and variability in vegetation structure (Chapter 2), allows an evidence-based description of structural variability. This allows the structural limits used by legislation to be evidence-based and defensible; a desirable property when these concepts are being argued in courts of law. I demonstrated another use in Chapter 6 by exploring the diversity of the inter-tidal communities and extending this information into an ecological context. When quantitatively identified communities are associated with mapping, their application becomes powerful in allowing possibilities of ecological comparisons on a national or global scale.

Notwithstanding these benefits, my results in Chapter 4 show there will always be communities included in the RE system that are identified by supervised techniques. In my

study area, this was because available qualitative data indicated a different community, but the lack of quantitative data prevented its inclusion in analysis. Lack of resources to access remote areas will always be a constraint in sparsely populated areas with a limited number of ecologists working in the field, and so communities detected by supervised methods may always exist. However, these are “known unknowns”, providing targeted areas for future work if resources become available.

With the greatest proportion of the natural vegetation in Queensland being sclerophyll communities (98.5%, Accad, Neldner, et al. (2019)) my new classification approach has wide applicability. However, the RE system includes rainforest communities and for these it specifies a classification approach with concepts based on structure, but the same dominance based criteria as sclerophyll communities (Neldner, Wilson, et al., 2019). However, there is widespread recognition that dominance concepts breakdown in species rich communities such as rainforest. Thus, there is a mis-match between the concepts and the criteria in the classification approach for identifying rainforest communities in Queensland. Defining an approach for these communities is an important next step in aligning the RE system with current best practice.

Recommending one set of class definition procedures across the whole state for non-rainforest communities is similar to the approach adopted by the United Kingdom but differs from that of the United States of America. The USA specifies using varying class definition procedures and assessing the results for convergence (Jennings et al., 2009), whereas the UK consistently uses TWINSPAN (Rodwell, 2006). This has resulted in a static classification system in the UK which is hard to update (Rodwell, 2018). It is possible the same may occur with the RE system and there may be a need to review my classification approach in the future.

My work has demonstrated the value of having the concepts and criteria of a classification approach explicitly defined (Chapter 5). I posit this is also one of the reasons the expert panel agreed with such a high proportion of the communities identified using the new class

definition procedures (96%), despite the large reduction in the number of communities (49%) indicating that the assumptions of ecologists regarding factors driving plant community difference were not supported (Chapter 4). A common experience when a new classification system is introduced is reluctance to accept it in favour of the established system (Wiser & De Cáceres, 2018), but my results suggest new classification exercises carried out in areas where there are previous classification systems should firstly define their existing concepts and criteria and either, choose plot-grouping techniques consistent with these, or be explicit and transparent in the rejection of them. Being specific and outlining the concepts and criteria of the new classification system will allow users to understand the differences between the old and the new, possibly enhancing the uptake of the new system.

Additionally, I would argue that explicit concepts and criteria are required for an Australian national vegetation classification system, an issue also recognised by (Gellie, Hunter, Benson, Kirkpatrick, et al., 2018). Currently there are two national vegetation classification systems (Executive Steering Committee for Australian Vegetation Information & Department of the Environment and Heritage, 2003; D. Keith & Tozer, 2017). Best practice suggests the goal should be a unified system (Bruelheide & Chytrý, 2000; De Cáceres et al., 2018; Jennings et al., 2009). If the concepts and criteria for both national and state vegetation systems were outlined using the framework developed by De Cáceres et al., (2015), it would allow an easier integration of the systems of individual States in Australia (Gellie, Hunter, Benson, Kirkpatrick, et al., 2018) into a national system and allow an easier integration of Australian vegetation information into an international system (Faber-Langendoen et al., 2014).

Barriers to implementation

Queensland is a large state with a sparse population and a limited number of ecologists working in the field of vegetation classification. For example, in north Queensland, across an area of 260 000 km^2 there are three ecologists involved in this work. This compares with European countries who have many more ecologists working in much smaller areas (Gellie,

Hunter, Benson, Kirkpatrick, et al., 2018; Mucina et al., 2016). Also, in contrast to Europe and other parts of the world the expertise in the RE classification system lies with the Government rather than Tertiary institutions, meaning it is more difficult to gain the skill set required to apply un-supervised class definition procedures. As a result of these factors, there are two major barriers to the implementation of my new classification approach. The first, identified earlier, is the perceived cost versus benefit of collecting data from detailed vegetation plots. If there are not enough detailed plots surveyed, identifying communities using my new classification approach is not possible and undermines the defensibility of the RE system. The second centres on the lack of institutional capacity; one of the considerations in the Government's decision to use agglomerative hierarchical clustering rather than fuzzy noise clustering as the plot-grouping technique. This lack of capacity also includes a limited understanding by other ecologists involved in curating and administering the RE system of the assumptions underlying the un-supervised class definition procedures in my new approach. Queensland is a large state dominated by savanna and rangeland systems with broad environmental gradients and my findings that ecologists perceive differences which do not represent landscape patterns in these situations (Chapter 5) means it is important my new classification approach is applied in a standardised manner across the State. To achieve this, it is imperative the Queensland Government invests in training the ecologists responsible for curating and administering the RE system otherwise an idiosyncratic application of my new classification approach will result. The other major reason why investment in training is imperative is to safeguard corporate knowledge and resilience. It is highly undesirable to have the expertise in the application of a state-wide government practice residing in one employee. Both strong managerial direction and investment in training are required to ensure the new classification approach is embedded into the RE system.

Conclusions

By determining quantitatively based class definition procedures for a system with already well-defined concepts and criteria my research gives a greater understanding of the issues surrounding vegetation classification and its application. The new classification approach developed in this thesis is being adopted as standard government practice. This fundamentally changes the way REs are defined in Queensland and aids the implementation of the Government's policy of evidence-based land management decisions. The RE system is tied to land use legislation at both the State and Federal levels and my research therefore has the potential to impact people's livelihoods and the biodiversity of the State. However, having vegetation communities which are the base-line level of the RE hierarchy based on quantitative analyses will ensure REs are more readily defensible and robust. This will likely instil greater confidence in the classification system in both legislators and end users.

Adopting my new classification approach will enhance the RE system's already wide use. As well as the current comparisons of spatial and temporal change of REs (Accad, Neldner, et al., 2019) statistical comparisons between vegetation communities at a cross-bioregion scale will become possible (Goodall, 1973). I anticipate communities identified using my new approach will aid investigations into such questions as the assumptions behind their use as surrogates for biodiversity (Sattler & Williams, 1999), the environmental drivers of the patterns of community distribution, or the phylogenetic diversity of communities. Importantly, it will also form statistically supported base-line data against which to measure the effects of future changes, such as climate and land use.

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Appendices

Appendix 1

Appendix 1.1 Guidelines for defining new regional ecosystem or vegetation community (V. J. Neldner et al., 2019)

A 1.1 New regional ecosystem

For a new regional ecosystem to be recognised, all requirements must be met, and at least one of the criteria conditions satisfied. These are in addition to the bioregion and land zone that are part of the regional ecosystem classification (section 2.3). It is expected in the majority of cases at least two criteria will be satisfied; that is, a change in landscape position will be reflected in a change in floristics or structure.

Definition: a vegetation community or communities[#] in a bioregion that is consistently associated with a particular combination of geology, landform and soil.
[#]Component vegetation communities may only be mappable at a scale larger than 1:100 000.

Caveats: The regional ecosystem framework is based on the 1:100 000 scale of mapping (Sattler and Williams 1999). When assigning land zones it is expected that geological or landsystem mapping at a comparable scale will be used.

A 1.1.1 Requirements

All requirements must be met.

 Area

Total pre-clearing area >100 ha, or if <100 ha then at least three distinct patches

 Information

Adequate information to assess the species, structure and landscape criteria is required. This will generally be in the form of a technical description derived from secondary or tertiary site data.

 Mappability

The regional ecosystem must be consistently mapped at regional scale.

 Equivalence check

Checked for equivalence in Regional Ecosystem Description Database (REDD)
http://www.ehp.qld.gov.au/ecosystems/biodiversity/regional-ecosystems/index.php

 Consultation

Other botanists/experts for bioregion consulted. Final endorsement required from bioregional technical committee via the bioregional coordinator, who is the senior author listed against each bioregion in REDD (Queensland Herbarium 2017a).

 Non-outlier

Regional ecosystem matches the description from an adjacent bioregion (that is, dominant species and land zone are equivalent), and has area in the bioregion of at least 1 000 ha or if less than 1 000 ha then occurs at least 50 km from existing bioregion boundary and occurs in more than two patches. If does not meet these area and/or distance requirements it is regarded as an outlier and coded with the regional ecosystem from the adjacent bioregion

A 1.1.2 Criteria

At least one of the criteria conditions must be met.

 Floristic

Dominant canopy species different from established regional ecosystems within the same bioregion and land zone;

or Combination of dominant and subdominant canopy species (species making up bulk of the biomass) different from established regional ecosystems. If the only floristic difference is in the subdominant canopy species, then at least one other criterion (structure or landscape) must also be satisfied;

or

If canopy matches established regional ecosystem, then a distinct, consistently present (>50% sites) shrub layer with at least 10% projective foliage cover, for example, *Eucalyptus populnea* woodland with *Eremophila mitchellii* shrub layer (RE 6.5.4), *Eucalyptus populnea* grassy woodland (RE 6.3.18). If the only floristic difference is in the shrub layer, then at least one other criterion (structure or landscape) must also be satisfied;

or

If canopy matches established regional ecosystem, then a distinct, consistently present (>50% sites) ground layer that is dominated by different species/growth form from established regional ecosystem, for example, *Acacia georginae* low open woodland with *Astrebla* spp. dominated ground layer (RE 4.9.14) or Acacia georginae tall open shrubland with *Triodia* spp. dominated ground layer (RE 4.5.7). If the only floristic difference is in the ground layer then at least one other criterion (structure or landscape) must also be satisfied;

 Landscape

Dominant species and vegetation description may fit established regional ecosystem, but occur on different landform and/or geological substrate from established regional ecosystem;

 Structural

Floristic description matches established association or sub-association, but the structural formation consistently occurs outside the structural range for the established association or sub-association, such as different Specht (1970) structural formation with a difference of at least 2 m in height and at least 5% projective foliage cover from established Specht formations, for example, *Eucalyptus tetrodonta, Corymbia nesophila* ± *Erythrophleum*

chlorostachys tall woodland (vegetation unit 2, Neldner and Clarkson 1995) and *Eucalyptus tetrodonta*, *Corymbia nesophila* woodland (vegetation unit 101, Neldner and Clarkson 1995). Generally, ecosystems are not differentiated on structure unless the landscape criteria or the ecosystem occupies a distinct geographical range.

Appendix 2

Appendix 2.1: Areas of low sampling adequacy by survey design on the Tertiary and igneous landscapes, Cape York Peninsula bioregion

Figure A2.2: Distribution of areas of Tertiary (LZ 5 on maps) and igneous (LZ 12 on maps) landscape which are <90%-similar to any site for each environmental variable. Because such large areas of both landscapes were >90%-similar to any site, for display purposes we show only areas with <90%-similarity. Areas on igneous landscape correspond largely with areas of rainforest which are not included in this study. These maps are indicative only. GIS layers are attached as KML files which will overlay Google Earth.

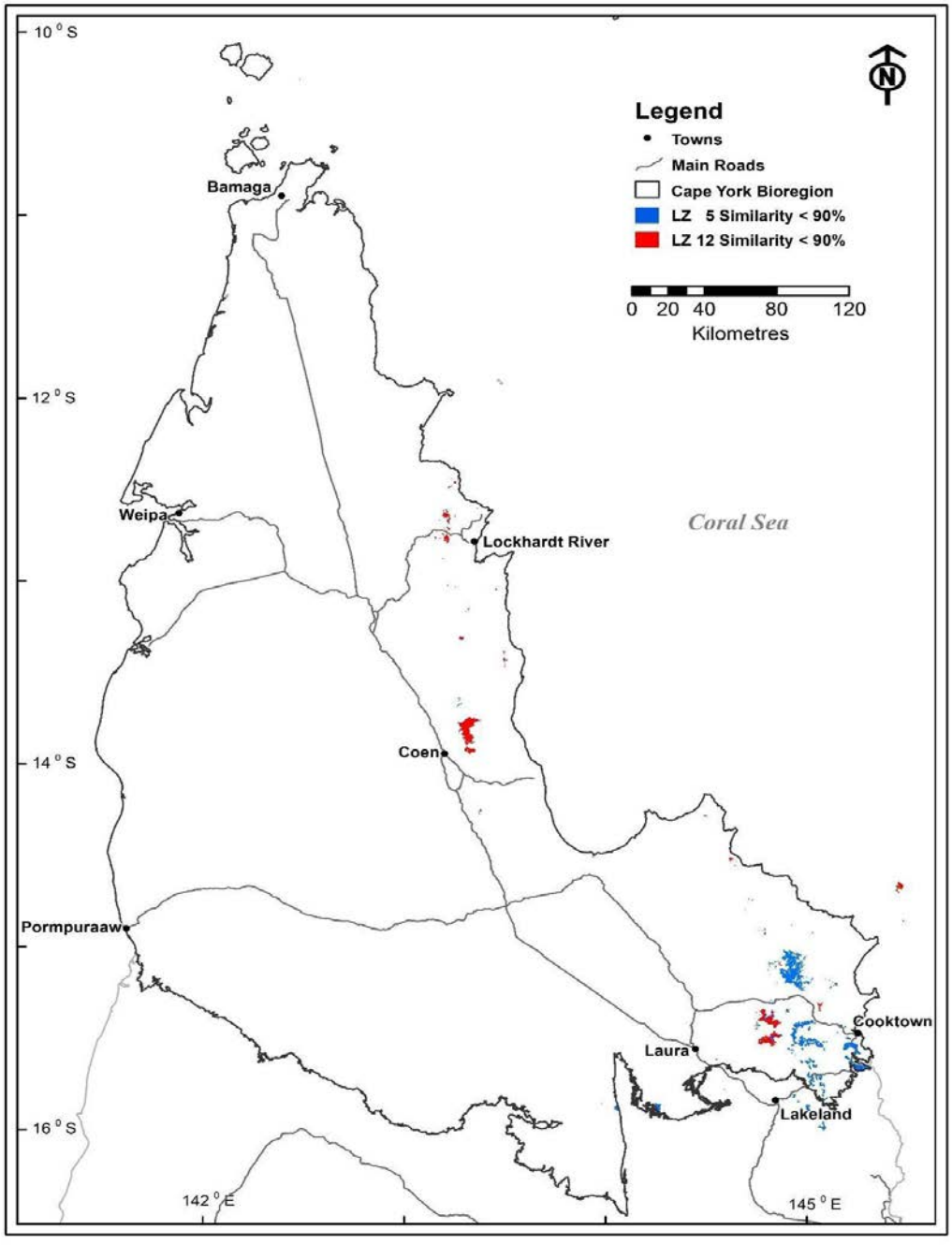


Figure A2.2a: Climate.

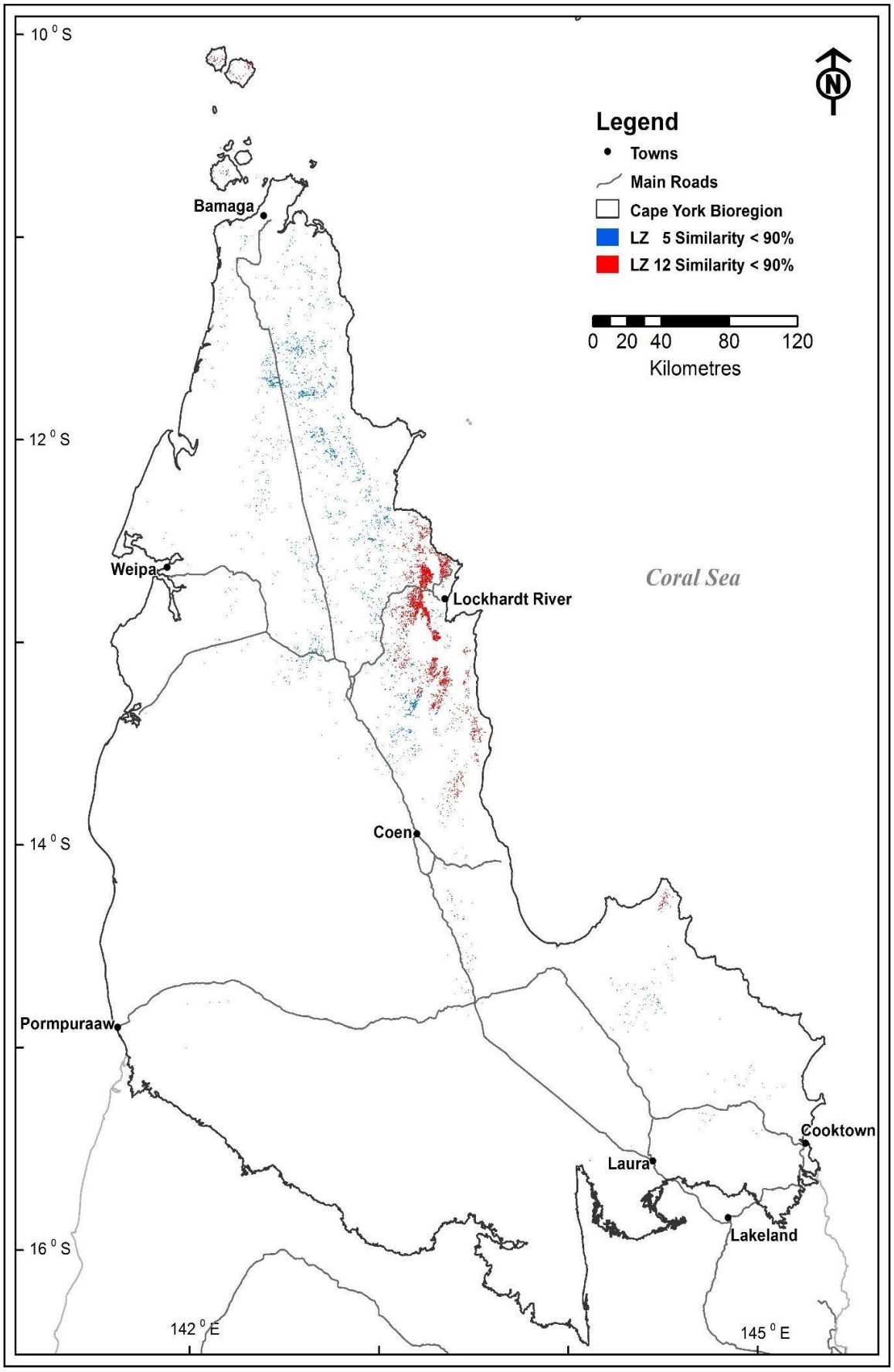


Figure A2.2b Woody vegetation density (represented by maximum persistent greenness)

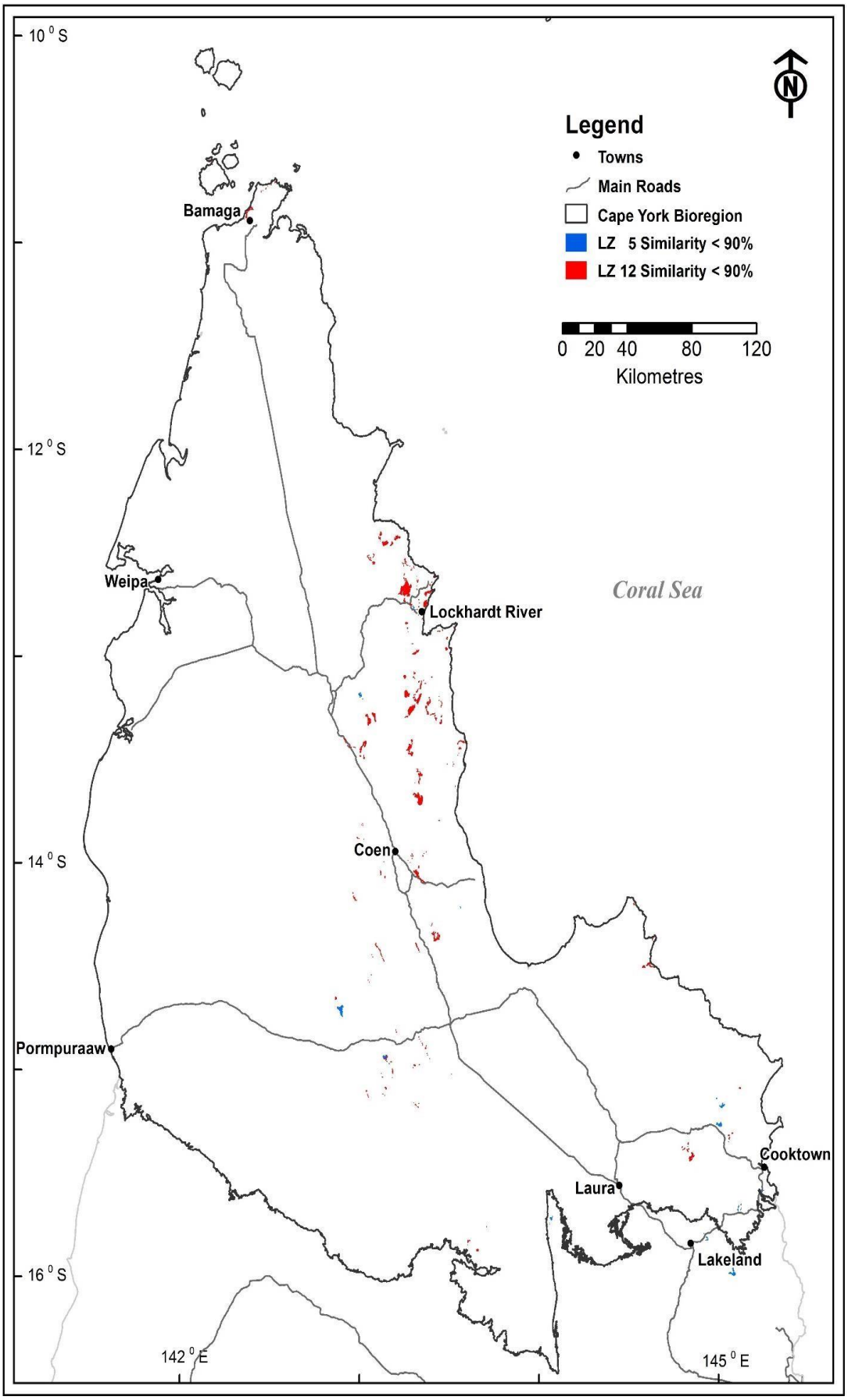


Fig 2.2c: Soil nutrient

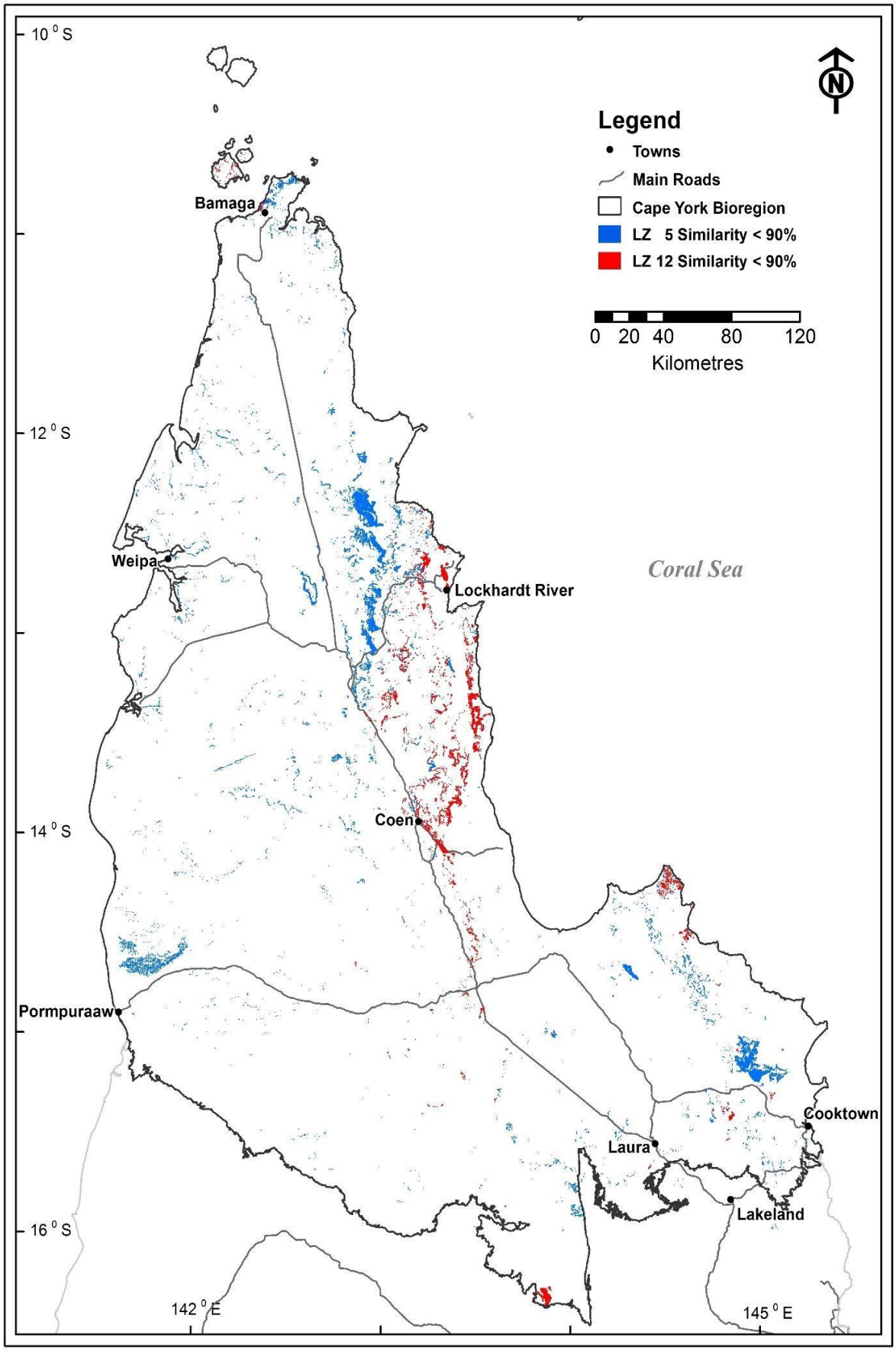


Figure2.2d: Soil structure

Appendix 2.2. Investigation into the correlations between persistent greenness index and climate variables.

We tested for correlations between climate variables and woody vegetation density using a 4-way ANOVA in the EXCEL stats package. Woody vegetation is represented by a maximum persistent greenness index (JRSRP 2017). The line-fit plots of woody vegetation density against each climate variable (Fig 6.1 – 6.4) provide a visualisation of the strength of correlation and the low predictability for woody vegetation. While there is a significant correlation between woody density and climate, the spread of actual woody vegetation values compared to expected values portrays the low predictability of woody vegetation density by climate (R^2 = 0.34)

Table A2.3 4-way ANOVA of woody vegetation density against climate variables.

Regression Statistics	
Multiple R	0.58
R Square	0.34
Adjusted R Square	0.34
Standard Error	13.00
Observations	1000

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	4	86136.3	21534.1	127.3	8.13988E-88
Residual	995	168254.7	169.1		
Total	999	254391			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	203.77	26.98	7.55	9.67745E-14	150.82	256.71

Temperature seasonality (C of V%)	-0.33	0.09	-3.51	0.0005	-0.51	-0.15
Mean moisture index of lowest quarter	1.75	0.39	4.50	7.63064E-06	0.99	2.52
Annual precipitation (mm)	0.02	0.01	3.23	0.0013	0.01	0.03
Annual mean temperature (^{0}C)	-2.76	0.88	-3.12	0.0018	-4.49	-1.02

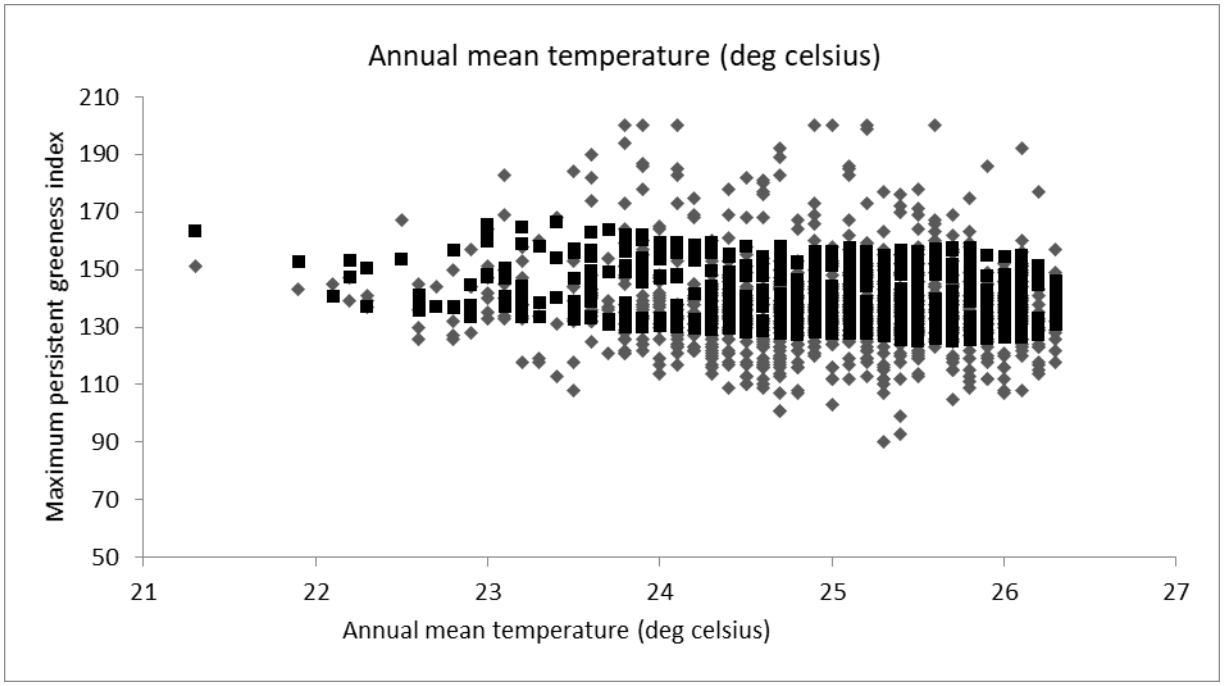


Figure A2.3.1. Line-fit plots of annual mean temperature against woody vegetation density (represented by maximum persistent greenness index). ◆ = actual maximum persistent greenness index at each observation point, ■ = predicted maximum persistent greenness index at each observation point

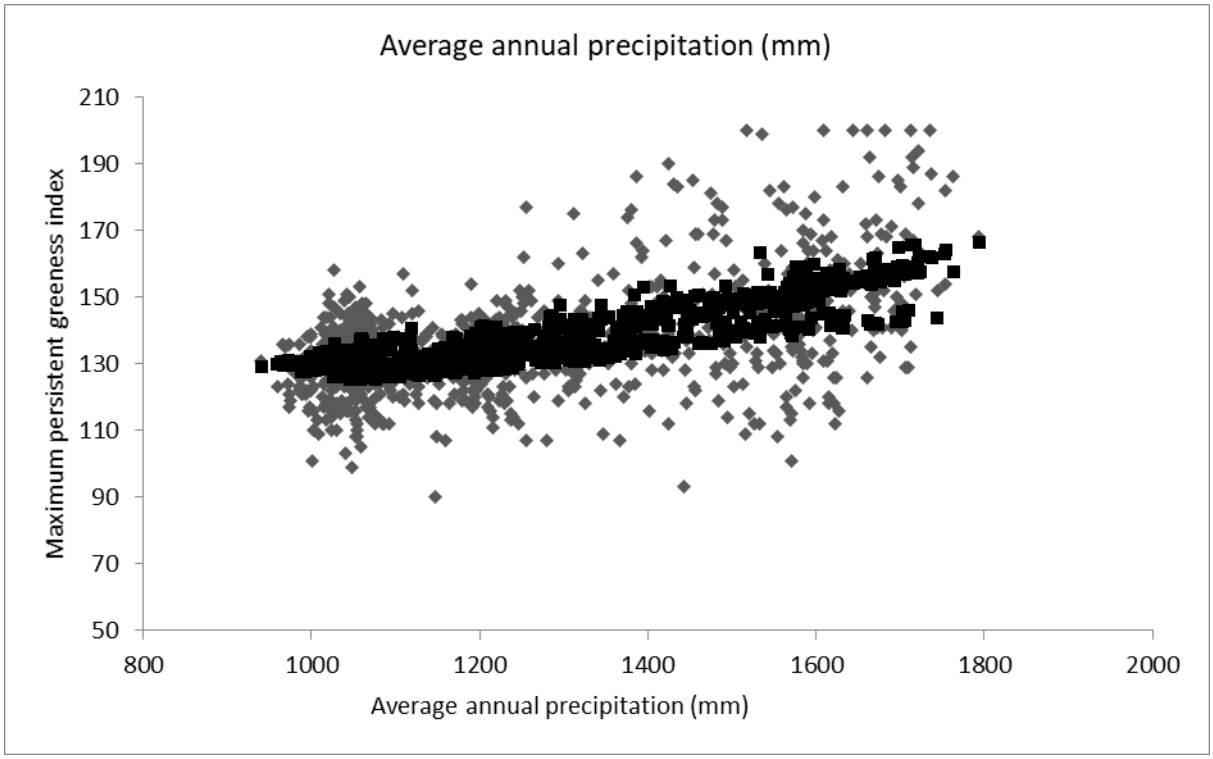


Figure A2.3.2. Line-fit plots of average annual precipitation against woody vegetation density (represented by maximum persistent greenness index). = actual maximum persistent greenness index at each observation point, = predicted maximum persistent greenness index at each observation point

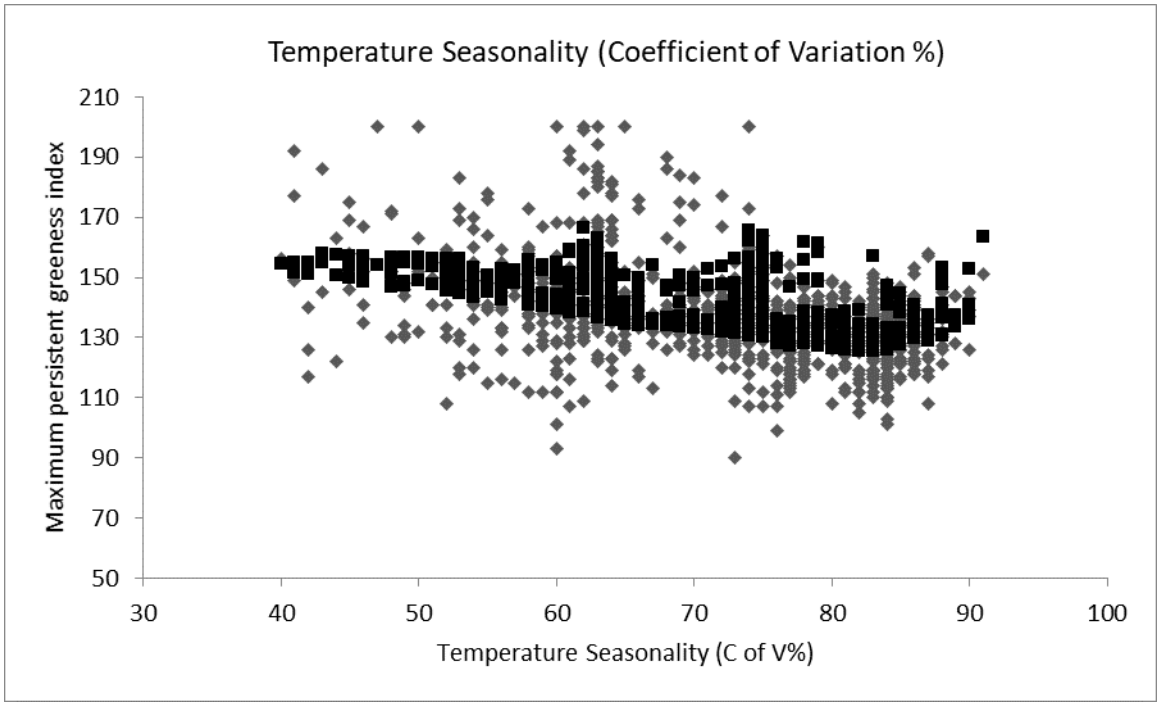


Figure A2.3.3. Line-fit plots of temperature seasonality against woody vegetation density (represented by maximum persistent greenness index). = actual maximum persistent greenness index at each observation point, = predicted maximum persistent greenness index at each observation point

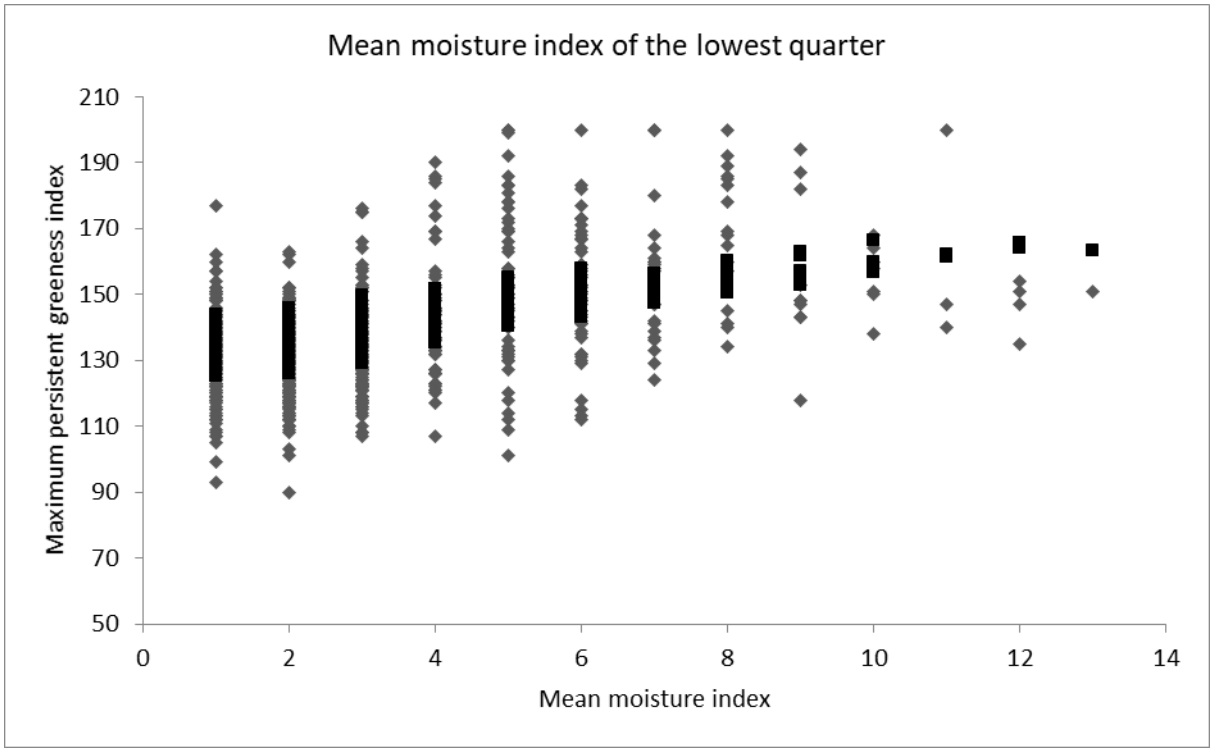


Figure A2.3.4. Line-fit plots of mean moisture index of the lowest quarter against woody vegetation density (represented by maximum persistent greenness index). ◆ = actual maximum persistent greenness index at each observation point, ■ = predicted maximum persistent greenness index at each observation point

References

(Joint Remote Sensing Research Project, 2017)

Appendix 3

Appendix 3.1 The Need for Standardised Frameworks in Vegetation Classification: A Literature Review

Abstract

Classification of vegetation communities is used in a broad range of applications on all continents. Techniques for data analysis to do this are extremely varied. The surveyed literature showed different analysis techniques produced different results, depending on the algorithm, irrespective of the size of the dataset, the abundance measure used, the synoptic information produced or the original purpose of the study. There is no general consensus among vegetation scientists on which techniques are preferable and numerous authors recommend that a standardised analysis approach is used and documented. De Cáceres and Wiser (2012) suggest a framework within which to do this. When the approaches used by the Australian states were analysed within this framework, I found standard approaches to data analysis vary greatly in detail and consistency; however standard classification schemes and vegetation survey techniques are widespread. Seven of the eight states or territories have yet to address most activities in the classification process in a standardised, documented manner. Documenting the approach to data analysis will ensure that comparisons between communities are robust. State-wide projects need to develop recommended approaches to data analysis as part of the suite of protocols that form a classification methodology.

Introduction

Observation shows different patterns of colour and texture repeating across the landscape which leads to the underlying assumption of vegetation classification; that patterns of vegetation are repeated (Whittaker, 1973b). Vegetation classification groups sites together to form clusters representing these patterns of colour and texture (D. Goodall, 1973). Clusters are strongly, but not exclusively based on floristic composition (Kent, 2012).

Vegetation classification underpins many land management decisions and much scientific research (Chytrý et al., 2011; De Cáceres & Wiser, 2012; Jennings et al., 2009; J. S. Rodwell, Pignatti, Mucina, & Schaminée, 1995; Sun et al., 1997). Decisions across large geographic areas become possible as do comparisons of change (Jennings et al., 2009; D. Walker et al., 2013) and when directly tied to mapping of vegetation communities it allows

detailed analysis of their extent (Accad, Neldner, Wilson, & Niehus, 2012; Executive Steering Committee for Australian Vegetation Information, 2003). The demand for vegetation inventory information is steadily increasing (Chytrý et al., 2011) because of its direct usefulness in applied questions (Wesche & von Wehrden, 2011) amongst other reasons. Delineation of vegetation associations also provides a base for ecological exploration of the patterns of species distribution, and spatial and temporal changes (K. R. Clarke & Gorley, 2006; Kent, 2012).

Vegetation classification techniques are of two types; subjective and objective (Kent, 2012; Whittaker, 1973b). Subjective techniques are based primarily on the researcher's ecological knowledge and expertise (Kent, 2012). Objective techniques were established with the development of computers and are based on statistical and mathematical models (D. Goodall, 1973; Kent, 2012). Both subjective and objective classification techniques are still in use across a range of vegetation types (for example Acebes et al. (2010); Kusbach, Long, Van Miegroet, and Shultz (2012); Zhu, Cao, and Hu (2006)) and study size areas (for example B. Anderson, Chiarucci, and Williamson (2012); Leite and Rodrigues (2008); Luther-Mosebach et al. (2012)).

National and state-wide vegetation classification projects change from 'early-stage' classification schemes with communities delineated by subjective techniques, to 'late-stage' classification schemes based on statistical analysis (Oliver et al., 2012; J. S. Rodwell, 2006). Part of this process includes development of a group of documented standardised components, including one or more of: a classification scheme; vegetation survey techniques; naming conventions and criteria for amalgamation or creation of associations within the classification scheme (Benson, 2006; Harris & Kitchener, 2005; Jennings et al., 2009; J. S. Rodwell, 2006; J. S. Rodwell et al., 1995). The objective of this review is to explore the need for identifying and documenting the process of statistical data analyses.

Methods and Definitions

The major search facilities for this review were 'OneSearch' and Thomson Reuters' 'Web of Knowledge', Scopus, Google Scholar and the Department of Science, Information Technology, Innovation and Arts (DSITIA) library. I also searched grey literature within Australian state and federal departments via government websites.

The keywords and phrases of 'vegetation classification', 'phytosociolog*' were the initial search terms. Keyword searches evolved as articles were retrieved and these new terms and phrases were used in subsequent searches. Supplementary searches were done

through cited literature lists of books and journal articles and some articles were sourced through recommendation from peers.

There were three broad search areas: history of vegetation classification, theoretical concepts and real world examples. These topics generated 126 references which were examined for this review.

In the context of this review these definitions of terms are used:

Queensland Methodology - a suite of techniques used in any given situation.

Classification scheme – the hierarchical divisions that group plant communities. For example the Queensland classification scheme is bioregion, land zone and vegetation type (Sattler & Williams, 1999), the U.S. National classification scheme is formation, division, alliance and association (Jennings et al., 2009).

Vegetation survey techniques – the way in which vegetation data is collected. This includes what attributes are recorded and how they are measured.

Analysis techniques - the techniques used to classify vegetation site data to produce the levels in the classification scheme.

Objective classification - statistical models and techniques used to determine vegetation associations. In peer reviewed literature it is also referred to as numerical classification.

Subjective classification - the techniques of classifying vegetation which rely on the researcher's ecological knowledge and experience without support from statistical analysis to test the robustness of the results. This is also referred to as expert-based methods.

The Need for Objective Classification

Subjective classification of vegetation is based on ecological knowledge to group sites of similar floristic composition (Kent, 2012). The most subjective technique involves the ecologist manually sorting site data, often from different surveys and using different measures and attributes, into groups that appear similar and which match the concepts of communities the ecologist has developed during field work (Benson, 2006; Blydenstein, 1967; Oliver et al., 2012).

A more structured approach to sorting site data into similar groups was developed by Braun-Blanquet (1928) and is still the basis of much vegetation survey and classification today (Kent, 2012). The process has been described in some detail by Westhoff and van der Maarel (1973) and summarised by Kent (2012). There are six main steps involving re-arrangement of the species by site matrix table into groups with similar species frequencies.

The end result is a synoptic table, traditionally with vegetation communities across the top and characterising species for each community down the side, populated by per cent constancy of each species in each community.

Subjective classification has some major limitations centered around its reliance on the ecologist. Field knowledge, experience and familiarity of the researcher with the communities in question (Kent, 2012) and the non-transparent character-weighting carried out during the process (L. Mucina, 1997) makes the variability of ecological expertise a major factor. Work done by Oliver et al. (2012) shows that expert based classifications are influenced strongly by dominant species with rare species being underweighted. Other problems with expert based methods revolve around psychological elements. External factors such as time pressure (Maule, Hockey, & Bdzola, 2000), time since a break or food (Danziger, Levav, & Avnaim-Pesso, 2011) or dynamics within an expert panel (Martin et al., 2012) can influence the decisions of experts. Cowan (2001) indicates that 4 ‘memory chunks’ (which in this context are ‘species’) may be the realistic limit for consideration by the ecologist’s short term memory, giving support to the idea that results are unlikely to be repeatable by multiple observers (Kent, 2012). In addition large datasets become unwieldy to sort by hand (Kent, 2012) and expert-based classifications are unable to produce statistically based indicator species, instead producing qualitative lists of important species (Harris & Kitchener, 2005; Oliver et al., 2012). Thus, subjective classifications generate hypotheses rather than test them (D. Goodall, 1973).

Objective classification has developed with the advent of computers (D. Goodall, 1973) and overcomes some of the problems outlined. By providing repeatability and consistency in the sense that the same techniques used with the same dataset will produce the same results (Kent, 2012; L. Mucina, 1997) hypotheses can be tested (D. Goodall, 1973), specifically those generated by subjective classifications (for example see Wesche and von Wehrden (2011); Willner (2011)). Objective classification allows statistical characterisations of vegetation communities (D. Goodall, 1973) and this information can be used in varying ways such as the first steps in key development (Willner, Tichý, & Chytrý, 2009), in comparison of temporal or spatial changes (K. R. Clarke et al., 2014; Kent, 2012) or in further applications of landscape classification (De Sanctis et al., 2013). Difficulties of handling large datasets are overcome (Kent, 2012) and ecologists are now capable of classifying datasets with thousands of sites (for example Chytrý et al. (2002); van Tongeren, Gremmen, and Hennekens (2008); Willner et al. (2009)). This capability has supported the development of databases that store large amounts of site data thus enhancing data sharing across jurisdictions and geographical areas (for example Chytrý et al. (2011); Rutherford, Mucina, and Powrie (2012); S. K. Wiser et al. (2011)).

Summary of Objective Classification Techniques.

A multiplicity of statistical techniques have been developed for objective classification (D. Goodall, 1973; Kent, 2012). They can be either hierarchical or non-hierarchical (K. R. Clarke & Gorley, 2006; De Cáceres & Wiser, 2012; Kent, 2012), with hierarchical classification based on the similarity (or the converse dissimilarity) between sites and non-hierarchical based on the concept that sites sit in multi-dimensional space. Hierarchical classification produces a dendrogram representing a hierarchy of the similarity between sites (D. Goodall, 1973; Kent, 2012). Non-hierarchical classification partitions the multi-dimensional space so that sites close to each other are considered a cluster (Kent, 2012). Clusters in both techniques are taken to represent vegetation communities at some level. One of the major considerations in the development of all techniques is the concept of robustness, where the results of the classification are not dependant on the underlying structure of the data (Kent, 2012).

Non-hierarchical techniques

Non-hierarchical techniques are less commonly used than hierarchical techniques. Based on set theory they were originally used for analysis of sociological data (De Cáceres et al., 2010) and L. Mucina (1997) describes the papers of Feoli and Zuccarello (1986, 1988, 1991), Dale (1988) and Moraczewski (1993) as being seminal in initiating its use in vegetation classification. Hard partitioning techniques give an absolute membership of a site to a cluster (De Cáceres, Schmidtlein, & Wiser, 2012; Kent, 2012). A disadvantage of this is needing a pre-defined number of clusters, something that is not generally known when classifying vegetation (Kent, 2012). Fuzzy partitioning takes prototypes and calculates the assignment of sites into clusters based on distance from the prototype (De Cáceres et al., 2010), giving sites a goodness-of-fit membership score to surrounding clusters. This is also useful for identifying transitional sites, acknowledging that plant species may be distributed along an environmental continuum (De Cáceres et al., 2010; L. Mucina, 1997). Although Banyikwa et al. use non-hierarchical techniques as far back as 1990 and the most recent examples include Mahecha, Martinez, Lange, Reichstein, and Beck (2009) and S. K. Wiser and De Cáceres (2012) it is still infrequently used.

Hierarchical techniques

The two predominant types of hierarchical classification are divisive and agglomerative clustering (De Cáceres & Wiser, 2012; Kent, 2012). Both of these techniques create groups of sites, based on their similarity to each other. Divisive clustering assumes all sites are similar and works in the same way as a dichotomous key, splitting sites into one of two

choices down a stepped hierarchy (Kent, 2012). Agglomerative clustering starts by assuming all sites are different and then groups sites together that are most similar. This process continues until all sites sit in one group (De Cáceres, Schmidtlein, et al., 2012; De Cáceres & Wiser, 2012; Kent, 2012).

Choosing an agglomerative technique involves a number of decisions. Firstly, there is the choice of transformation to apply to the data. This has the effect of smoothing out and balancing the contribution of each species to the calculation of the similarity coefficient (K. R. Clarke, Chapman, Somerfield, & Needham, 2006). The choices range from none, which gives no weighting to rare species, through to presence / absence in which rare species are given equal weight to abundant species (K. R. Clarke & Gorley, 2006; Kent, 2012). Second is the choice of similarity coefficient, of which three are most commonly used by vegetation scientists; the Jaccard coefficient, the Steinhaus (Sorensen/Czekanowski/Bray-Curtis) coefficient and Euclidean distance coefficient. The final consideration is the sorting strategy which is the technique used to calculate how samples are progressively allocated to cluster groups (Kent, 2012). There are three common choices here also; nearest neighbour, furthest neighbour or group average linkage (K. R. Clarke & Gorley, 2006; Kent, 2012).

In the surveyed literature agglomerative hierarchical clustering was the most common statistical approach, with 31 of the 73 articles using it. Within these, the combination of square-root (or log +1) data transformation, Bray-Curtis coefficient and UPGMA sorting strategy was the most common (26 articles). This combination is described by Belbin and McDonald (1993), K. R. Clarke and Gorley (2006) and Kent (2012) as the optimal statistical approach for species datasets and is supported by the research of Wesche and von Wehrden (2011) in their work in southern Mongolia.

The predominant divisive clustering technique still in widespread use is TWINSPAN (M. O. Hill, 1979). It is used as the primary analysis technique in 11 of the 52 reviewed articles, in another three to compare results with agglomerative cluster analysis as a way of result validation and is used in papers appearing as recently as 2012 (Kusbach et al., 2012; Luther-Mosebach et al., 2012). Despite criticisms which centre around the fact that the analysis is dependent on a “dominating primary gradient” and may not show other existing gradients (Belbin & McDonald, 1993), that the derived ‘pseudo-species’ are artificial in their combination of species by mid-point of class, and that the description of the technique was unpublished, researchers find that it gives ecologically meaningful results (Kent, 2012).

Ordination analysis may be used on its own for classification (D. L. Lewis & Phinn, 2011), but is mostly used in conjunction with cluster techniques. It allows a visual check indicating whether cluster results are arbitrary or real (complimentary analysis) (Kent, 2012). For

ecological datasets, Non-metric Multi-dimensional Scaling is the most suitable algorithm as it is shown to be the non-parametric analysis method robust enough to deal with the often sparse data (K. R. Clarke & Gorley, 2006; Kent, 2012).

Validating results

Once clusters are produced by a statistical model, knowing which clusters are valid can be problematic; where on a dendrogram does the ecologist decide that the clusters are forming valid vegetation communities (De Cáceres et al., 2010)? Fourteen different methods and combinations were used in the surveyed literature to do this. These can be broadly divided into comparing the results of different techniques with each other (51 of the 73 articles surveyed) or comparing information inherent in the cluster (11 articles).

Complimentary analysis comparing ordination and hierarchical clustering was the most frequently used of all validation methods (15 of the 73 articles reviewed). This was followed by comparing results from different statistical models (11 articles). Other methods included testing the statistical significance of cluster groups (Bell, 2013; Oliver et al., 2012) or using the same technique across different datasets (Koci et al., 2003; van Tongeren et al., 2008) thus testing the robustness of the technique. Where vegetation mapping is the primary aim of the project, assessment of the recognition of cluster groups on remotely sensed imagery is used (Bedward et al., 1992; Neldner, Fensham, Clarkson, & Stanton, 1997; Neldner & Howitt, 1991; Penn, Sutton, & Monro, 2004).

Using information derived from the cluster group to validate vegetation communities relies on species-based measures. Fidelity, based on constancy and frequency, tests the strength of association of an individual species to a cluster (Barkman, 1989; Bruelheide, 2000; Tichý, 2002). Indicator Species Values (produced from Indicator Species Analysis) is another index for evaluating the strength and significance of a species' association with a cluster group (Dufrêne & Legendre, 1997) and recently De Cáceres, Legendre, Wiser, and Brotons (2012) suggest that Indicator Species Analysis based on a group of species may be more effective than that calculated with a single species. Willner (2006) suggests using a slightly modified Barkman (1989) degree of fidelity based on calculating the Total Cover Value of an individual species, giving criteria for 'good' diagnostic species, diagnostic species and non-diagnostic species.

All validation methods are tools (Kent, 2012). Recognition of the vegetation types represented by cluster groups is also emphasised as the most important consideration by a number of other authors (D. Goodall, 1973; Whittaker, 1973a). In 13 of the surveyed articles

this was the only validation method and it was explicitly used in conjunction with a statistical or mathematical technique in 40 of the surveyed articles. It was implicitly used in all others.

Considerations in Choosing Objective Classification Techniques

The choice of which techniques to use can be influenced by a number of factors. These are based around the effects different algorithms have on results, the abundance measures used in analysis, the synoptic information produced and the original purpose of the classification.

Algorithms

The same dataset can produce different results depending on the algorithms used (De Cáceres et al., 2010), and this can occur in either large or small datasets. Wesche and von Wehrden (2011) applied five agglomerative and one divisive algorithm to the same species-poor dataset from across southern Mongolia. 1231 sites included vegetation types of annual desert vegetation, forests and steppe grasslands, with semi-arid vegetation and steppe grasslands being most common. Differences in clusters were produced by the assumptions of the algorithms rather than differences within the vegetation types. With a contrasting dataset size of 17 in a 27ha area M. Anderson and Clements (2000) had a similar result when assessing the outcomes of four algorithms. The subjective classification had recognised three open forest types, agglomerative hierarchical clustering produced three different open forest types and divisive clustering again produced three types of yet different composition from the previous two.

Different algorithms can produce similar communities but at different levels of divisions. Baruch (2005) analysed data from 37 sites across four types of savanna vegetation. While divisive and agglomerative clustering produced generally similar results, these were at different levels of hierarchy in the dendrogram. Burgman and Thompson (1982) also found that two agglomerative clustering algorithms gave similar results at higher levels of division representing the landscape level (for example wetlands and sandstone ranges) but different communities were produced at lower levels due to different emphases in the algorithms.

Dataset structure

Dataset structure may influence the outcome of an objective classification technique. Bruelheide and Chytrý (2000) found the same algorithm used on two datasets, with known similar vegetation communities, failed to produce the equivalents. Different indicator species

were produced by the second datasets due to outlier sites, which were found to represent an environmental gradient not present in the first. Chytrý et al. (2002) found that the size of the dataset affected the determination of diagnostic species, with them being more difficult to statistically identify in smaller datasets than in large.

Abundance measures

Using different abundance measures from the same sites also provides different results. In areas where there are a large number of sites, European researchers have found that presence/absence gives a more robust outcome than abundance measures. Willner et al. (2009) show that in forest and dwarf shrub lands the most important factor in determining the efficacy of an analysis technique and fidelity measure is whether presence/absence or abundance data is used. While presence/absence is most often used and acceptable for species-rich communities, in species-poor communities it was found important to use abundance-based measures. B. Anderson et al. (2012)and Q. Guo and Rundel (1997) both found that different abundance measures used to calculate dominance affect the ability to distinguish between tree communities when using species-abundance distributions. It is possible this may also be true for classifications using dominance.

Synoptic information

The synoptic information for communities produced by different techniques differs in both type of information and level of detail. Software such as PRIMER-E, using agglomerative techniques, produces % contribution of species to clusters, while the divisive software TWINSPAN produces negative and positive preferential species (with a cover value) (M.O. Hill & Šmilauer, 2005; Kent, 2012). These different types of information about cluster character influences the communities recognised and the information used to describe a community thus making it difficult to compare across studies (Bruelheide & Chytrý, 2000; Torello-Raventos et al., 2013; D. Walker et al., 2013). This finding is supported by results from Kusbach et al. (2012) who analysed 157 sites in forested and non-forested vegetation in northern United States of America to produce diagnostic species information. Divisive clustering algorithms produced different faithful species compared with agglomerative algorithms.

Purpose

Some techniques are more useful than others depending on the original purpose of the classification. Asking the question of how well does the algorithm results represent the existing subjective classification scheme Wesche and von Wehrden (2011) found the most

widely supported combination of Bray-Curtis similarity coefficient and UPGMA sorting strategy produced results most similar to the subjective classification. Divisive clustering produced similar results at the higher levels of division but not at the lower divisions. When asking how well the algorithm recognises environmental gradients the results from different algorithms approximated each other.

There is no general agreement on which technique for vegetation classification is most suitable (Koci et al., 2003; Wesche & von Wehrden, 2011) and the number of techniques available complicate their application. Vegetation data and classification schemes are increasingly used across state and national boundaries and large geographical areas (Bruelheide & Chytrý, 2000; Chytrý et al., 2011; Jennings et al., 2009; D. Walker et al., 2013). It is therefore important to consider the assumptions and ramifications of each technique when choosing which one to use.

The Goal of Consistency

The ultimate outcome of a vegetation classification exercise is the production of a set of conventions for allocating new sites to a classification scheme (S. K. Wiser & De Cáceres, 2012). The need to standardise this exercise has been recognised by a variety of administrations at various scales and is a common goal among vegetation scientists (Benson, 2006; L. Mucina, 1997; Sun et al., 1997; Torello-Raventos et al., 2013). The classification exercise involves a number of different components that combine together to form a methodology. These components can be summarised as: standardised approaches for vegetation survey, data analyses and naming conventions; documentation of the process for describing new units and a classification scheme, which outlines the hierarchy and level of the communities.

Different administrations have different components contributing to their standardised methodologies but naming conventions for vegetation communities is the most common (for example Executive Steering Committee for Australian Vegetation Information (2003); Jennings et al. (2009); Neldner et al. (2012); (J. S. Rodwell, 2006)). The United States of America has a methodology documenting most of the components mentioned (Jennings et al., 2009), as does Britain (J. S. Rodwell, 2006) and Canada ((Ponomarenko & Alvo, 2000) cited in (Jennings et al., 2009)). European countries are working together to develop a European Vegetation Survey (Chytrý et al., 2011) and common classification scheme (J. S. Rodwell, 2006) and naming conventions are formalised to the extent of being proscribed within a Latin based taxonomic nomenclatural framework (Weber, Moravec, & Theurillat, 2000). D. Walker et al. (2013) recently recognise the need for a pan-arctic standard vegetation classification methodology. Australia has a classification scheme however none

of the other components outlined (Executive Steering Committee for Australian Vegetation Information, 2003).

Data analysis, as shown, is a complex process but in the cited literature is one of the least defined components of any methodology. To overcome this De Cáceres and Wiser (2012) propose a framework within which to consider data analysis, breaking the classification exercise into two major tasks. The first is determination of the communities and the second is the consistent assignment of new sites to those communities.

The first task, determination of communities, is broken into four major activities:

1) Membership determination – grouping of sites into clusters of highest similarity representing communities.

2) Characterisation – producing synoptic information summarising and describing the communities.

3) Validation – determining whether a classification result is suitable for the original purpose.

4) Naming – labelling each community in a methodical way that fits the rules of the classification scheme.

Defining membership rules is the final important outcome from the first task. Membership rules are statements allowing new vegetation sites to be allocated to communities within the classification scheme in a consistent manner (task 2).

De Cáceres and Wiser (2012) argue for clear distinction to be made in published standards between the different activities in the first task because of the interlinked nature of the outcomes. Clarity between the membership determination and validation activities is particularly important, as is documentation of the membership rules of the final vegetation communities.

Scrutiny of the states in Australia within this context shows most states have a number of components of a standardised classification methodology (table 1). All except the A.C.T. have developed their own classification scheme. Six of the eight states and territories have developed standard vegetation survey techniques; four have outlined standards for accepting new vegetation associations and all have standardised naming conventions, if not explicitly defined then implicit in the classification scheme.

However, when approaches to data analysis are considered it becomes clear the detail varies from state to state. Casting the existing approaches into the De Cáceres and Wiser (2012) framework shows that the four activities have not been systematically addressed by

most states (table 1). Three recommend using objective data analysis techniques. Of these New South Wales addresses all four of the De Cáceres and Wiser (2012) activities (Sivertsen, 2009), the Northern Territory suggests procedures for addressing most of the activities, however not that for determining characteristic species (Brocklehurst, Lewis, Napier, & Lynch, 2007) and Queensland recommends using objective classification but does not recommend an approach or any techniques. The other states have not recommended an approach to objective data analysis.

Documentation of membership rules is as equally varied. New South Wales has them implicitly included in an identification key for individual vegetation communities. Tasmania also has rules implicit in a key but for map units not communities. The Northern Territory is in the process of developing community membership rules. The other states do not have any documented.

Conclusion

Objective classification allows repeatable, consistent and robust statistical information about community structure and composition (Kent, 2012). Although subjective classifications can provide important information about changes in community extent (Accad et al., 2012), objective classification schemes allow further questions to be investigated about change in and across communities (Chytrý, Tichý, Hennekens, & Schaminée, 2013). This can open up the potential to derive predictive models for future change scenarios about alterations in community structure and composition with regard to issues such as climate change or land management regime change. Questions such as the direction of compositional movement of grasslands and shrubby woodlands, how fast the C_3/C_4 species composition balance is changing or what the effect of changing saline incursions from storm surges is on salt-fringing communities may be investigated.

Progress towards standardisation of classification schemes and the development of methodologies by Australian states is substantial however approaches to data analyses are the least developed component (table 1). Documenting a standard approach to data analyses overcomes some of the complications surrounding objective techniques and, if done within the framework proposed by De Cáceres and Wiser (2012), is transparent and the outcomes clearly defined. Assignment of new sites into the classification scheme will be consistent, increasing the reliability and robustness of vegetation community information and the usefulness of the classification schemes.

Table 1. Components of vegetation classification methodologies of the Australian states (in the context of the De Cáceres and Wiser (2012) framework). E.M. = Ecological meaningfulness.

State	**Classification Scheme**	**Vegetation survey methodology**	**Data Analysis activities**					**Standards for recognising new units documented**	**Naming Conventions recommended**
			Membership determination *(Notes)*	**Attributes used for analysis**	**Validation**	**Characterisation**	**Membership Rules documented**		
N.S.W (Sivertsen, 2009)	Plant community types	Yes	Yes – expert + numerical – compare hierarchical & non-hierarchical	Determined by sub-set analysis of data	Heterogeneity analysis and E.M	Characteristic and diagnostic species - Fidelity measure, Indicator Values	Yes – implicit in key	Under development	Yes – although not documented specifically
N.T. (Brocklehurst et al., 2007)	Definitive Vegetation Type (using NVIS hierarchy)	Yes	Statistical (hierarchical agglomerative) & Expert based	Structure and floristics as per NVIS hierarchy	Testing with homogeneity & gradient analysis & E.M	Dominant or diagnostic species *(but technique of determining diagnostic species not defined)*	Under development	Under development	Yes – NVIS hierarchy
Qld (Neldner et al., 2012)	Regional Ecosystems	Yes	Expert based *(recommends using objective techniques – but not specified)*	EDL & Structure & species dominance	E.M	Occurrence & Cover *(although not a formal fidelity measure)*	No	Yes	Yes
W.A.	Formation; Alliance;	Yes	Not specified	EDL & Structure & dominant or	Not specified	Not specified	No	No	Partly

(Beard, Beeston, Harvey, Hopkins, & Shepherd, 2005)	Association (Vegetation types)			diagnostic species					
S.A (Heard & Channon, 1997); (Department for Environment and Heritage, 2006)	Vegetation Associations or Types – not explicitly specified	Yes	Not specified	Structure and floristics as per NVIS hierarchy	Not Specified	Dominant / codominant of overstorey, canopy, emergent (up to 3 spp.)	No	No	Yes (NVIS)
Vic (Department of Sustainability and Environment, 2013)	Ecological Vegetation Communities	Not specified	Not specified	Structure & species dominance	Not specified	In individual reports though not easily accessible on web – appears to be fidelity measure	No	No	No
Tas (Harris & Kitchener, 2005)	Vegetation Communities - map unit (with floristic communities underneath)	Yes	Not specified	Not specified	Not specified	Not specified	Not for floristic communities. But have key for map units.	Yes	Yes

ACT (ACT Government, 2013)	Not specified – but included in the NSW plant community types	Not specified	Not specified	Not specified	Not specified	Not specified	Not specified	Not specified	Not specified

Appendix 3.2 Supporting information to the paper Addicott, E. et al. 'When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savannas'. *Community Ecology* 19, 67-76. doi:10.1556/168.2018.19.1.7

Appendix 3.2.1 Data Analysis tools and methods for each question

*Tests used to measure the quality of the classifications.

	What are the effects on classification outcomes of:	
Analytical tools	Question 1: Removing rare species at different levels of contribution to TFC?	Question 2: Weighting species cover by different measures of vegetation layer height?
Determining cluster division	SIMPROF and ISA	SIMPROF
	(ISA produces Indicator Species (IS)	
	Comparison point = ALL dataset	Comparison point = NoHeight dataset
Tests in common		

1	Change in patterns of similarity between sites - 2STAGE in PRIMER-E	
2	Changes in clustering patterns - changes in proportion of clusters per formation and sites per cluster - Fisher's exact test ($p<0.05$)	
3*	Ability of classification to predict all species cover - generalised linear models and Akaike's Information Criteria (Lyons et al 2016)	
Specific investigations	**(Q1 – per formation)**	**(Q2 – whole dataset)**
	Changes in species richness per site (Margalef's Index) - unpaired t-test	*Which layers drove the classifications?
	Changes in species evenness per site (Pielou's Index) - unpaired t-test	
	*Changes in proportions of total-to-useful Indicator Species - Fisher's exact test	

Appendix 3.2.2 Classification dendrograms as a result of removing species based on % contribution to total foliage cover

Each value of β in the flexible-β clustering method produced the same clusters in the subsets as the full species pool (ALL dataset). However, Bray-Curtis coefficient with UPGMA clustering amalgamated clusters with each subset, which I show in figure below.

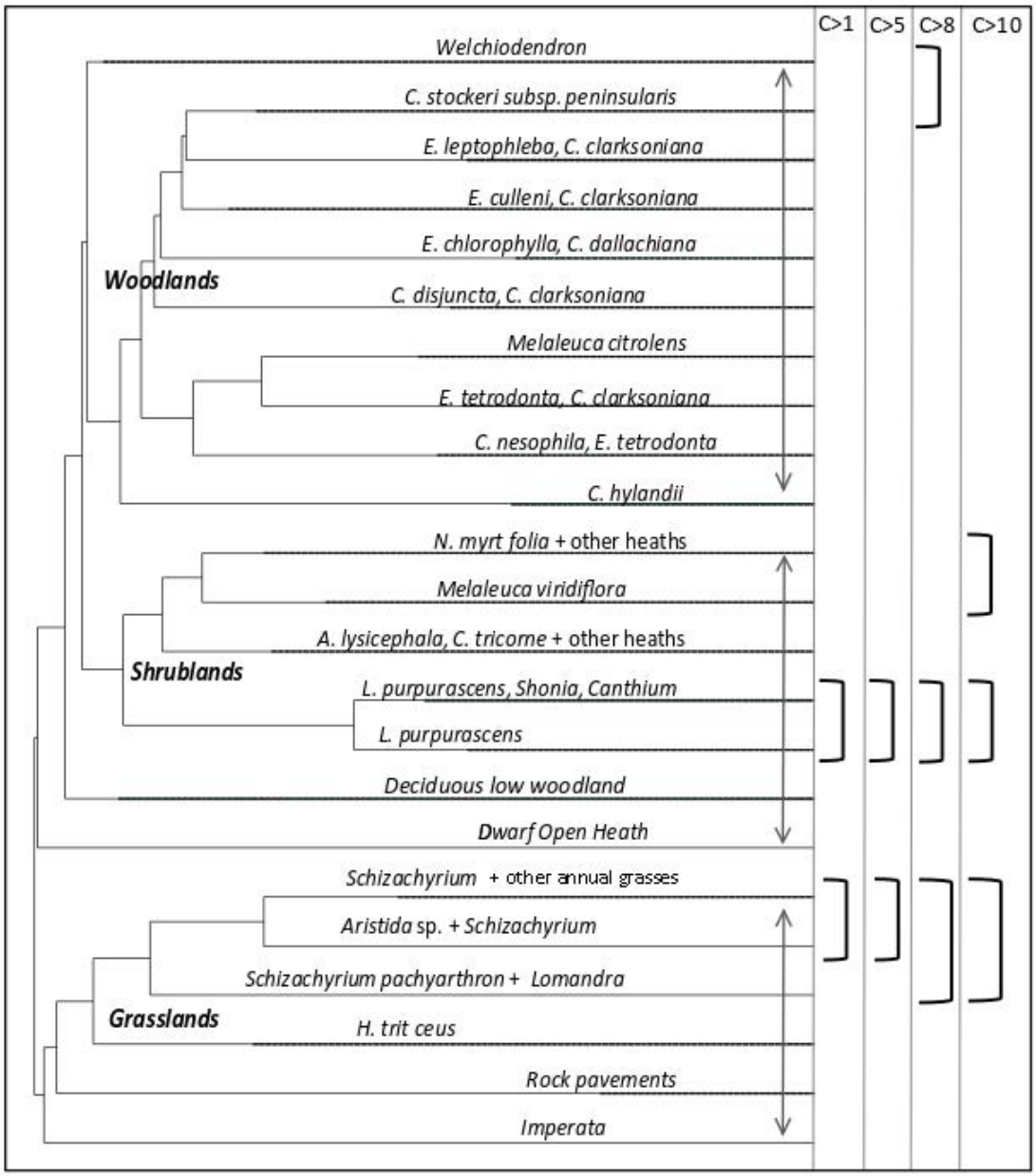


Figure A3.2: Clusters resulting from datasets with species removed based on % contribution to total foliage cover. Clusters are derived from Bray-Curtis coefficient and UPGMA linkage. Brackets indicate clusters that are amalgamated in different subsets. Clusters are labelled with species contributing >10% to similarity of sites in cluster. Species datasets: ALL = full species pool, C>1 = only species contributing >1% to total foliage cover, C>5 = species >5%, C>8 = species >8%, C>10 = species contributing >10% to total foliage cover.

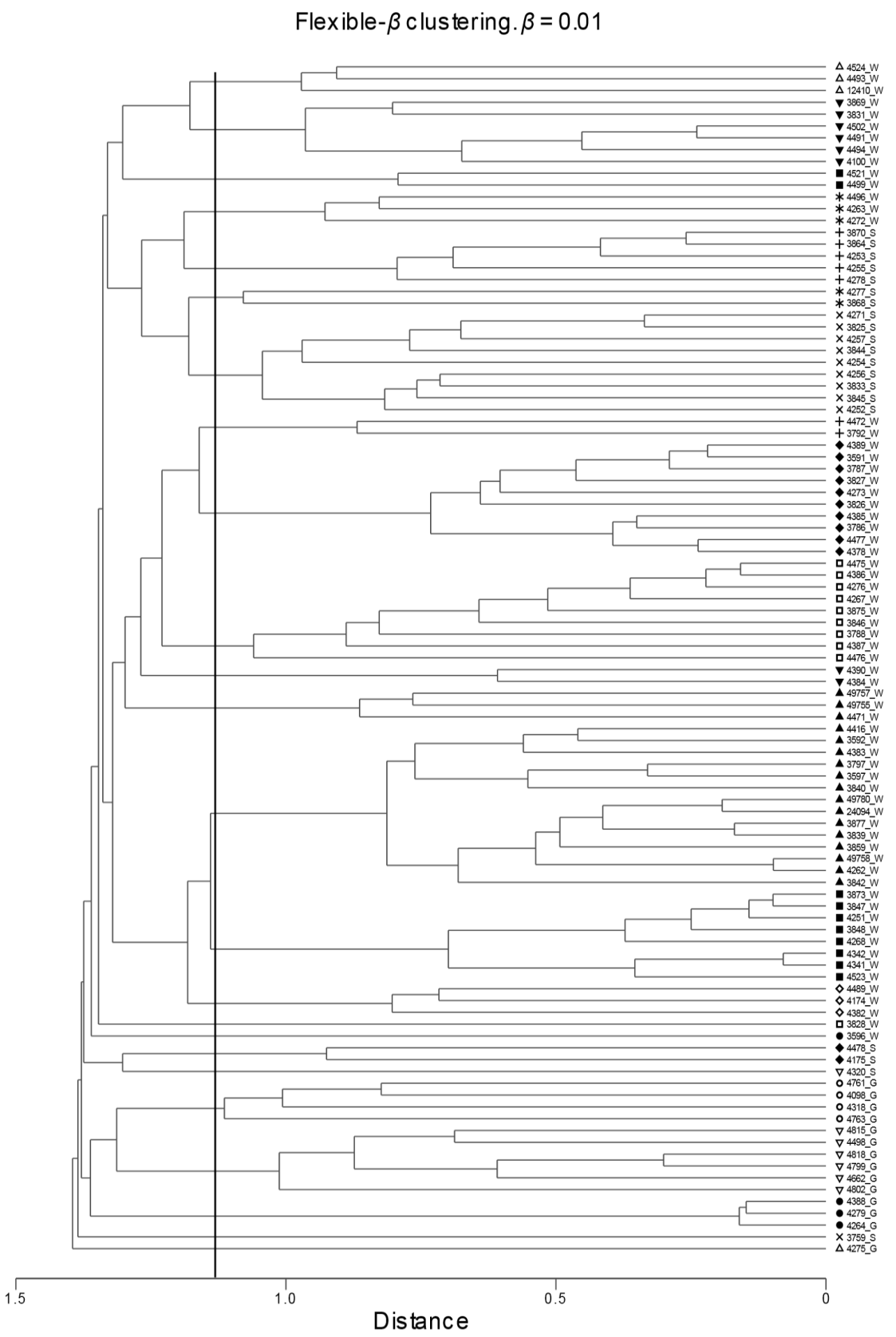


Figure A3.2. Clusters resulting from flexible-β clustering with optimal β value (0.01) chosen to maximise the cophenetic correlation between the distance matrix and the dendrogram. The optimal cluster grouping is indicated by the vertical black line (24 groups).

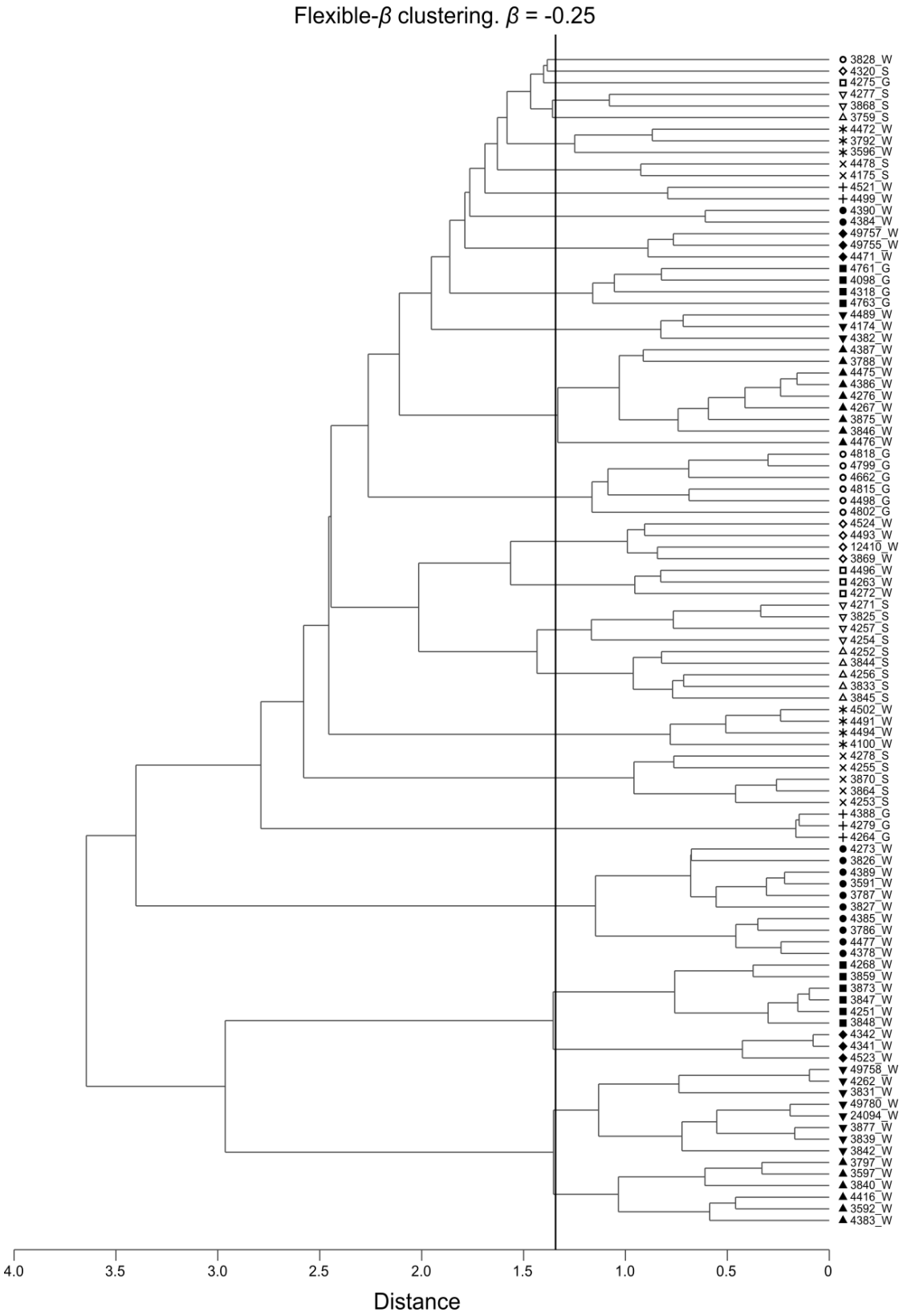
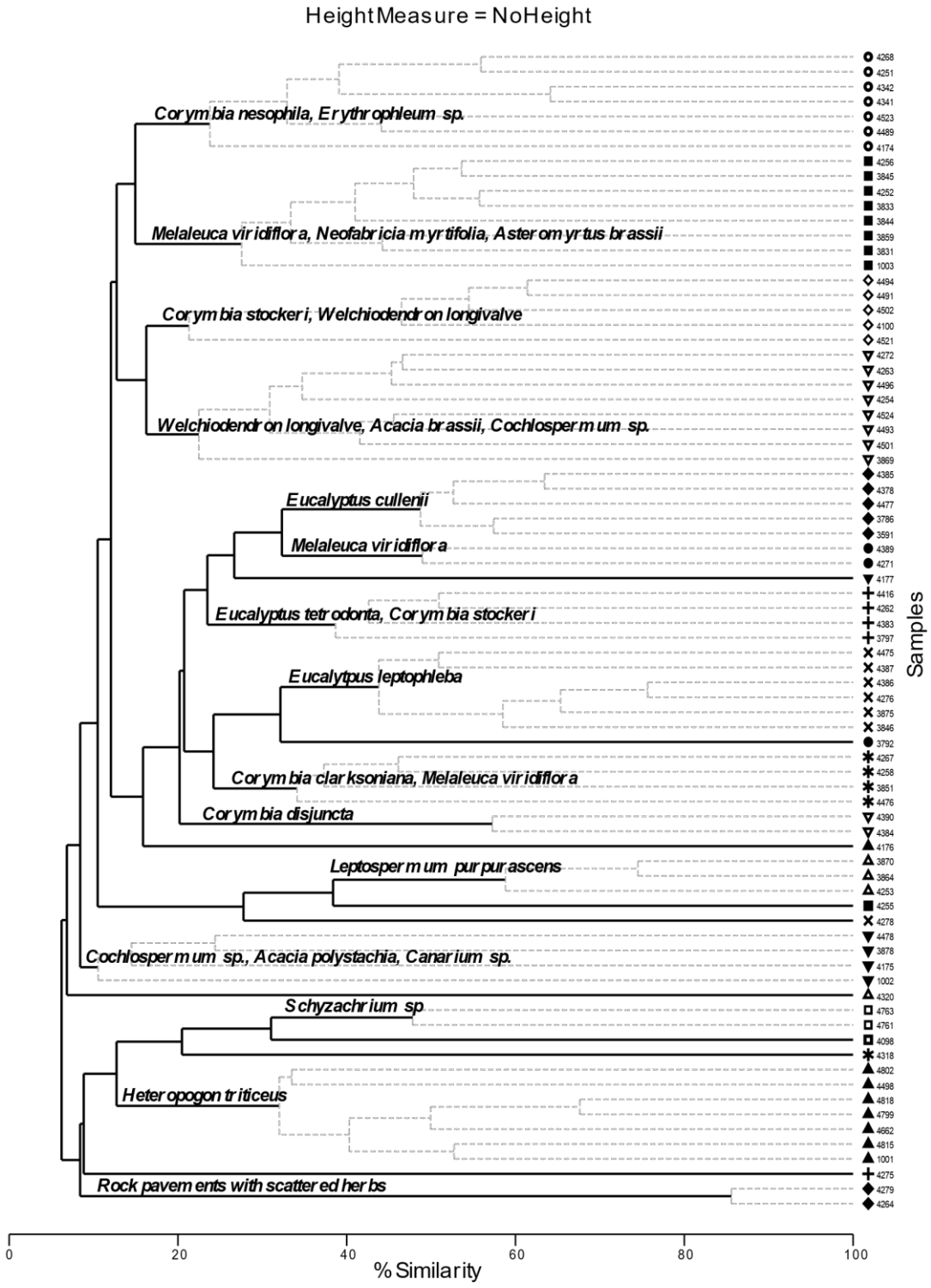
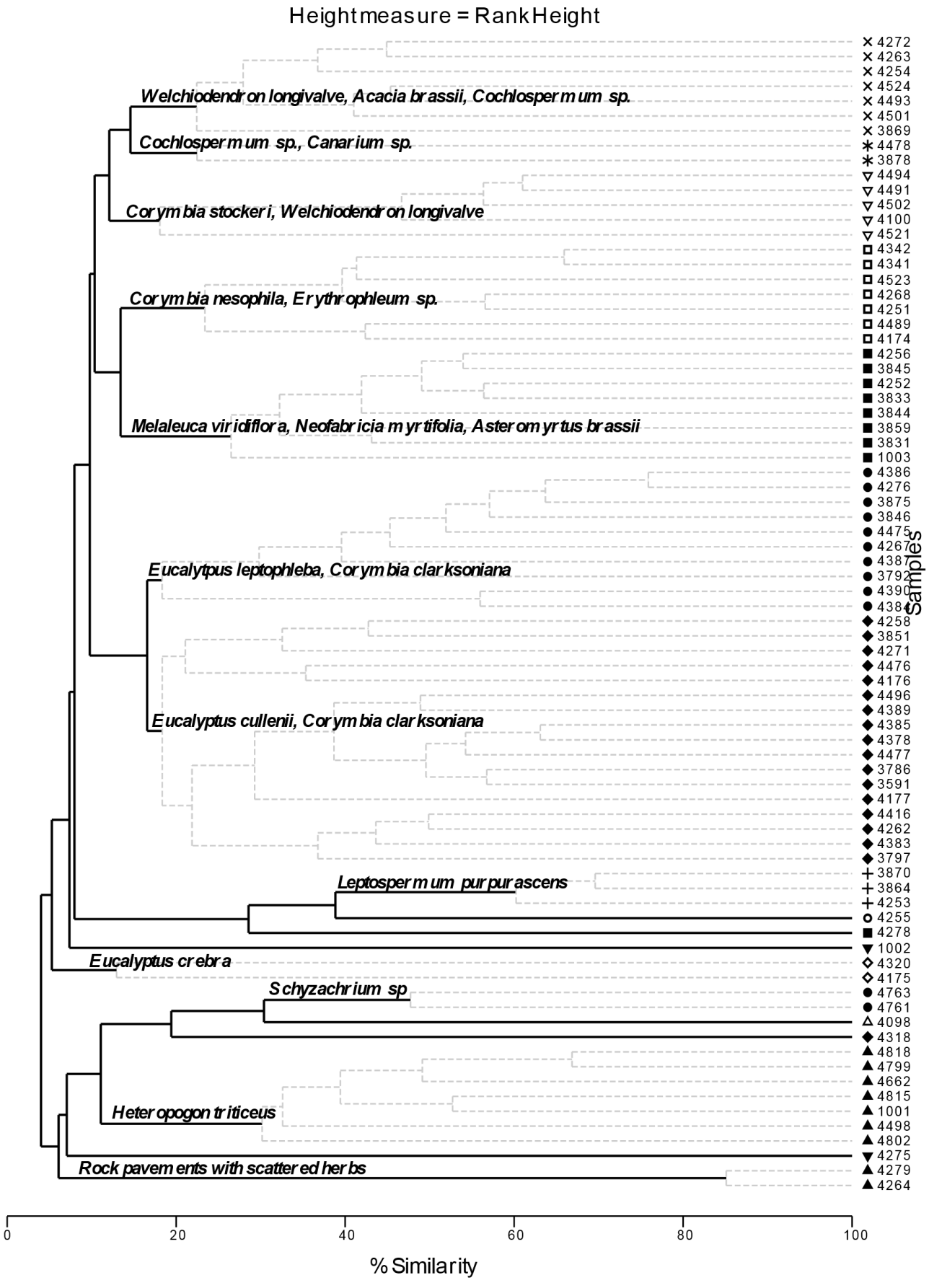


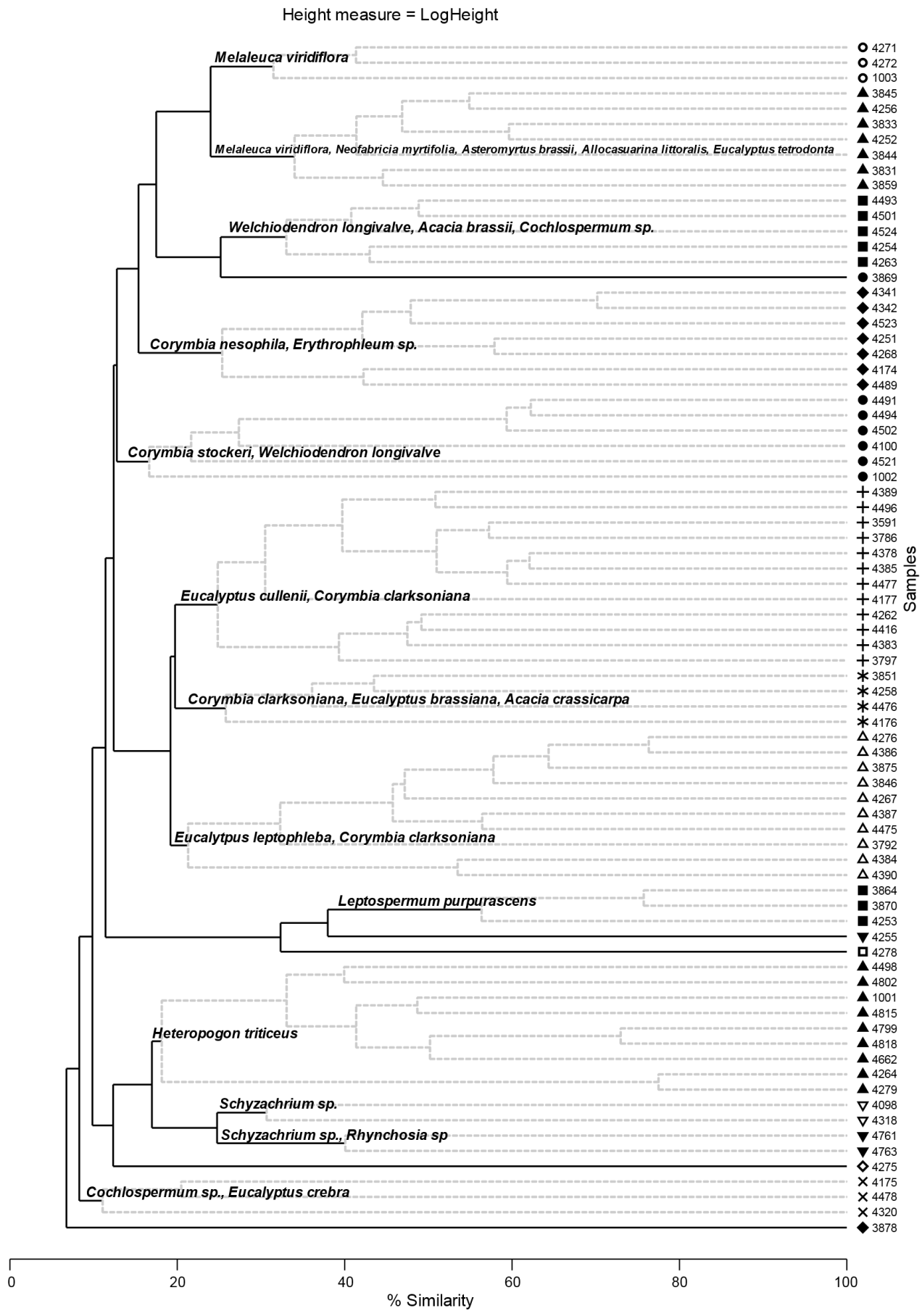
Figure A3.2. Clusters resulting from flexible-β clustering with β value set to -0.25 (a value widely used in ecology and classification). The optimal cluster grouping is indicated by the vertical black line (26 groups).

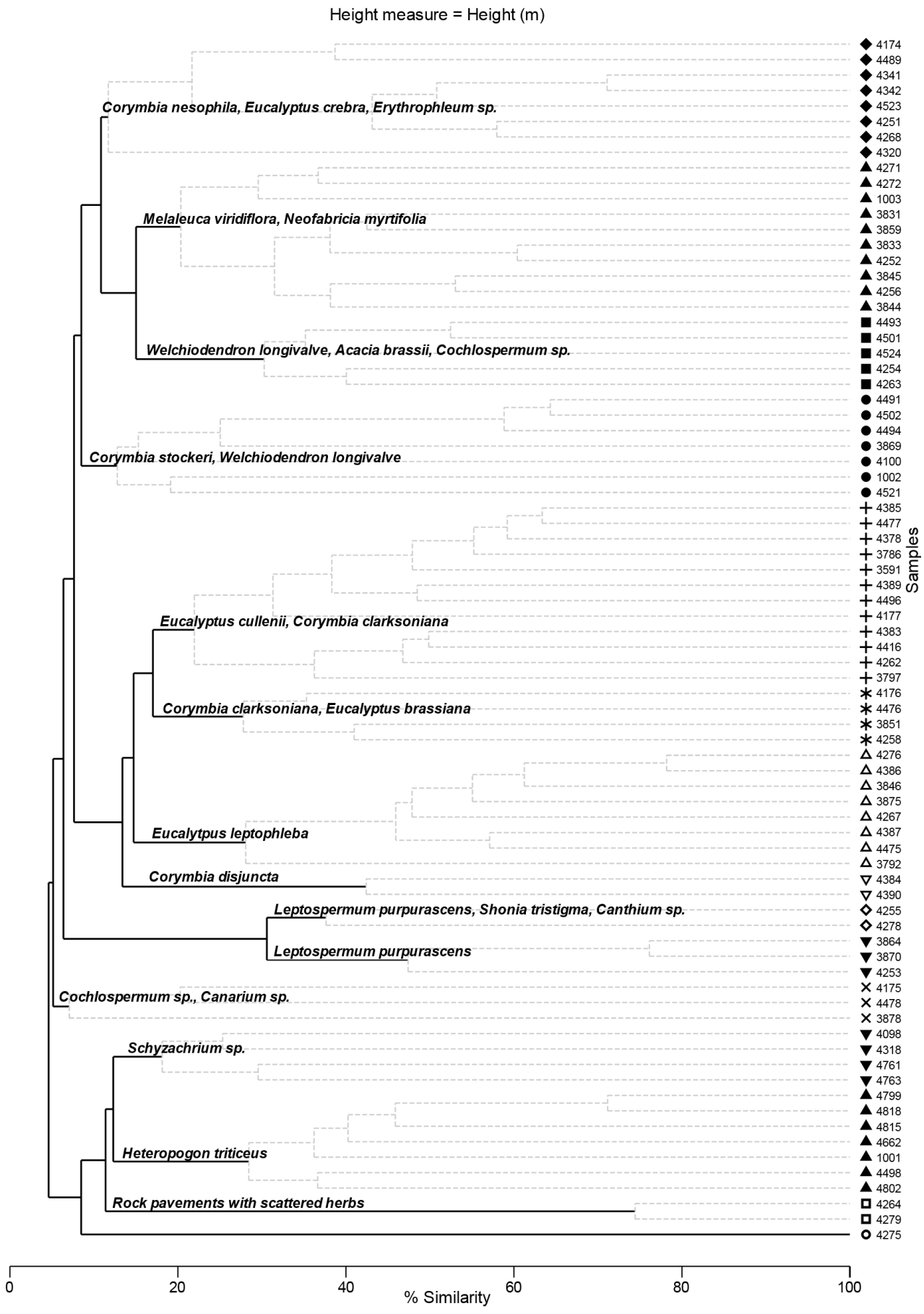
Appendix 3.2.3 Dendrograms of each classification after incorporating different height measures

Dendrogram titles indicate the height measure. Height = height in meters, LogHeight = $\log_{10}(x+1)$ of height, RankHeight = expert weightings of layers, NoHeight = no height included, foliage cover only. Clusters are labelled with species contributing >10% to the similarity between sites in a cluster.




Heightmeasure = RankHeight
Welchiodendron longivalve, Acacia brassii, Cochlospermum sp.
Cochlospermum sp., Canarium sp.
Corymbia stockeri, Welchiodendron longivalve
Corymbia nesophila, Erythrophleum sp.
Melaleuca viridiflora, Neofabricia myrtifolia, Asteromyrtus brassii
Eucalytpus leptophleba, Corymbia clarksoniana
Eucalyptus cullenii, Corymbia clarksoniana
Leptospermum purpurascens
Eucalyptus crebra
Schyzachrium sp
Heteropogon triticeus
Rock pavements with scattered herbs
4272
4263
4254
4524
4493
4501
3869
4478
3878
4494
4491
4502
4100
4521
4342
4341
4523
4268
4251
4489
4174
4256
3845
4252
3833
3844
3859
3831
1003
4386
4276
3875
3846
4475
4267
4387
3792
4390
4384
4258
3851
4271
4476
4176
4496
4389
4385
4378
4477
3786
3591
4177
4416
4262
4383
3797
3870
3864
4253
4255
4278
1002
4320
4175
4763
4761
4098
4318
4818
4799
4662
4815
1001
4498
4802
4275
4279
4264
Samples
0
20
40
60
80
100
% Similarity


Height measure = LogHeight
Melaleuca viridiflora
Melaleuca viridiflora, Neofabricia myrtifolia, Asteromyrtus brassii, Allocasuarina littoralis, Eucalyptus tetrodonta
Welchiodendron longivalve, Acacia brassii, Cochlospermum sp.
Corymbia nesophila, Erythrophleum sp.
Corymbia stockeri, Welchiodendron longivalve
Eucalyptus cullenii, Corymbia clarksoniana
Corymbia clarksoniana, Eucalyptus brassiana, Acacia crassicarpa
Eucalytpus leptophleba, Corymbia clarksoniana
Leptospermum purpurascens
Heteropogon triticeus
Schyzachrium sp.
Schyzachrium sp., Rhynchosia sp
Cochlospermum sp., Eucalyptus crebra
4271
4272
1003
3845
4256
3833
4252
3844
3831
3859
4493
4501
4524
4254
4263
3869
4341
4342
4523
4251
4268
4174
4489
4491
4494
4502
4100
4521
1002
4389
4496
3591
3786
4378
4385
4477
4177
4262
4416
4383
3797
3851
4258
4476
4176
4276
4386
3875
3846
4267
4387
4475
3792
4384
4390
3864
3870
4253
4255
4278
4498
4802
1001
4815
4799
4818
4662
4264
4279
4098
4318
4761
4763
4275
4175
4478
4320
3878
Samples
0
20
40
60
80
100
% Similarity


Height measure = Height (m)
Corymbia nesophila, Eucalyptus crebra, Erythrophleum sp.
Melaleuca viridiflora, Neofabricia myrtifolia
Welchiodendron longivalve, Acacia brassii, Cochlospermum sp.
Corymbia stockeri, Welchiodendron longivalve
Eucalyptus cullenii, Corymbia clarksoniana
Corymbia clarksoniana, Eucalyptus brassiana
Eucalytpus leptophleba
Corymbia disjuncta
Leptospermum purpurascens, Shonia tristigma, Canthium sp.
Leptospermum purpurascens
Cochlospermum sp., Canarium sp.
Schyzachrium sp.
Heteropogon triticeus
Rock pavements with scattered herbs
4174
4489
4341
4342
4523
4251
4268
4320
4271
4272
1003
3831
3859
3833
4252
3845
4256
3844
4493
4501
4524
4254
4263
4491
4502
4494
3869
4100
1002
4521
4385
4477
4378
3786
3591
4389
4496
4177
4383
4416
4262
3797
4176
4476
3851
4258
4276
4386
3846
3875
4267
4387
4475
3792
4384
4390
4255
4278
3864
3870
4253
4175
4478
3878
4098
4318
4761
4763
4799
4818
4815
4662
1001
4498
4802
4264
4279
4275
Samples
0
20
40
60
80
100
% Similarity

Appendix 3.2.4 Equations for indices used

1. Margalef's index *(d):* I used this to calculate species richness as it is independent of sampling size (K. R. Clarke et al., 2014). The index achieves this by incorporating the total number of individuals *(N)*, to adjust for capturing more species as the sample size increases (K. R. Clarke et al., 2014).

$$d = (S-1) / \log N$$

where *S* is total number of species

2. Peilou's evenness index (*J'):* is specifically a measure of the evenness of species abundance. This is derived from the Shannon diversity index

$$H' = -\sum[(p_i) \times ln(p_i)]$$

where p_i is the proportion of the total count arising from the *i*th species.

The Pielou's evenness index is based on the maximum diversity, that is a situation where all species are equally abundant. It is derived from the Shannon index and can expressed as

$$J' = H' / H_{max} = H' / log\ S$$

where *S* is the total number of species (Magurran, 2004, K. R. Clarke et al., 2014)

Clarke, K. R., Gorley, R. N., Somerfield, P. J., & Warwick, R. M. (2014). *Change in marine communities. An approach to statistical analysis and interpretation.* https://doi.org/1

Magurran, A. E. (2004). *Measuring Biological Diversity*. Oxford: Blackwell Publishing Ltd.

Appendix 3.3 Synoptic tables

Synoptic table of groups from agglomerative hierarchical clustering with characterising species. Characterising species calculated on Fidelity with all groups standardised to equal size and $p < 0.05$

Agglomerative hierarchical clustering Percentage synoptic table with fidelity (Phi coeff. C) (23 columns)

No. sites	1	2	1	2	5	1	2	9	4	1	2	13	16	3	2	7	3	14	3	5	4	1	6
Endiandra glauca	25.2	---	---	---	---	---	---	---	---	---	---	---	0.3	---	---	---	---	---	---	---	---	---	---
Cochlospermum gillivraei	---	27.6	---	---	---	---	---	3.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Terminalia subacroptera	---	16.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Diospyros compacta	---	14	---	---	---	2.2	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Gardenia	---	13.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	2.5	---	---	---	---	---
Memecylon pauciflorum	---	11.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Imperata cylindrica	---	---	27.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Heteropogon contortus	---	---	21.2	---	---	0.3	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Mnesithea rottboellioides	---	---	19.5	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Rock pavement	---	---	---	18.3	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Heteropogon triticeus	---	---	---	---	18.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Sarga plumosum	---	---	---	---	12.2	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Aristida	---	---	---	---	---	18.4	1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Schizachyrium	---	---	---	---	---	15.5	7.2	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Ectrosia	---	---	---	---	---	---	12.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Eriachne	---	---	---	---	---	---	10.5	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Polycarpaea spirostylis	---	---	---	---	---	---	10	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Welchiodendron longivalve	---	---	---	---	---	---	---	22.7	0.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia brassii	---	---	---	---	---	---	---	14.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Corymbia stockeri subsp. *peninsularis*	---	---	---	---	---	---	---	---	36.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia tessellaris	---	---	---	---	---	---	---	---	---	---	31.6	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia nesophila	---	---	---	---	---	---	---	---	---	---	---	---	34.3	---	---	---	---	---	---	---	---	---	---
Asteromyrtus brassii	---	---	---	---	---	---	---	---	---	---	---	---	14.6	---	---	---	---	---	---	---	---	---	---
Corymbia disjuncta	---	---	---	---	---	---	---	---	---	---	---	---	---	25.1	---	---	---	---	---	---	---	---	---
Eucalyptus chlorophylla	---	---	---	---	---	---	---	---	---	---	---	---	---	---	27.7	---	---	---	---	---	---	---	---
Eucalyptus leptophleba	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	30.9	---	---	---	---	---	---	---
Dendrolobium umbellatum var. umbellatum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	11	---	7	---	---	---	---	---
Eucalyptus brassiana	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	35.9	---	---	---	---	---	---
Corymbia clarksoniana	---	---	---	---	---	---	---	---	---	---	---	4.8	---	---	---	8.8	21.7	5.9	---	---	---	---	---
Calophyllum sil	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	13.8	---	---	---	---	---	---
Eucalyptus cullenii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	24.4	---	---	---	---	---
Melaleuca citrolens	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	23.3	---	---	---	---
Melaleuca foliolosa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	12.6	---	---	---	---
Leptospermum purpurascens	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	29.5	---	---	---
Asteromyrtus lysicephala	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	12.4	---	---
Terminalia arenicola	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	19.6	---
Eugenia reinwardtiana	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	19.6	---
Acacia leptostachya	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	19.5	---
Dodonaea viscosa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	17	---
Premna serratifolia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	13.5	---
Melaleuca viridiflora	---	---	---	---	---	---	---	---	---	---	---	---	2.6	---	---	---	---	---	---	---	---	---	16.6
Eucalyptus tetrodonta	---	---	---	---	---	---	---	---	---	---	---	22.9	12.6	---	---	---	---	---	---	---	---	---	---
Stenanthemum argenteum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.9	---	---

Hibbertia	---	---	---	---	---	---	---	---	---	---	---	---	1.8	---	---	---	---	---	---	---	5.3	---	---
Dodonaea	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	3.5	---	---
Acacia humifusa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	6.9	---	---
Lithomyrtus obtusa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.7	---	---
Labichea nitida	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.9	---	---
Choriceras tricorne	---	---	---	---	---	---	---	---	---	---	---	---	3.3	---	---	---	---	---	---	---	6	---	---
Jacksonia thesioides	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.8	---	---
Neofabricia myrtifolia	---	---	---	---	---	---	---	---	---	---	---	---	6.6	---	---	---	---	---	---	---	1.6	---	---
Petalostigma banksii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	7.3	---	---	---	---
Alphitonia pomaderroides	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia polystachya	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	2	---	---	---	---	---	---
Allocasuarina littoralis	---	---	---	---	---	---	---	---	---	---	---	---	8.1	---	---	---	2.8	---	---	---	---	---	---
Hibbertia banksii	---	---	---	---	---	---	---	---	---	---	---	---	3	---	---	---	---	---	---	---	---	---	---
Corymbia dallachiana	---	---	---	---	---	---	---	---	---	---	---	---	---	---	8.4	---	---	---	0.5	---	---	---	---
Acacia crassicarpa	---	---	---	---	---	---	---	---	---	---	---	---	1.9	---	---	---	8.6	---	---	---	---	---	---
Acacia platycarpa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Sersalisia unmackiana	---	---	---	---	---	---	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Smilax australis	---	---	---	---	---	---	---	1.3	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia flavescens	---	---	---	---	---	---	---	---	---	---	---	---	1.9	---	---	---	---	---	---	---	---	---	---
Euroschinus falcatus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	8.5	---	---	---	---	---	---
Banksia dentata	---	---	---	---	---	---	---	---	---	---	---	---	4.2	---	---	---	---	---	---	---	---	---	---
Acacia rothii	---	---	---	---	---	---	---	---	---	---	---	---	8.5	---	---	---	---	---	---	---	---	---	---
Coelospermum reticulatum	---	---	---	---	---	---	---	---	---	---	---	---	4.6	---	---	---	---	---	---	---	---	---	---
Lophostemon suaveolens	---	---	---	---	---	---	---	---	---	---	---	---	0.4	---	---	---	---	---	---	---	---	---	---
Abutilon albescens	---	---	---	---	3.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Opilia amentacea	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Parsonsia rotata	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.9	---	---

Livistona muelleri	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Thaumastochloa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Lamprolobium fruticosum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Tephrosia juncea	---	---	---	---	---	---	---	---	3.2	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Helicteres semiglabra	---	---	---	---	---	---	---	5.6	---	---	---	---	3.2	---	---	---	---	---	---	---	---	---	---
Xanthorrhoea johnsonii	---	---	---	---	---	---	---	---	---	---	---	---	5.1	---	---	---	---	---	---	---	---	---	---
Grevillea parallela	---	---	---	---	---	---	---	---	---	---	---	---	3.8	---	---	---	---	---	---	---	---	---	---
Dolichandrone heterophylla	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Colubrina asiatica	---	---	---	---	3.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia midgleyi	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Breynia	---	---	---	---	---	---	---	---	---	---	---	---	0.7	---	---	---	---	---	---	---	---	---	---
Persoonia falcata	---	---	---	---	---	---	---	---	---	---	---	---	5.3	---	---	---	---	---	---	---	---	---	---
Tabernaemontana pandacaqui	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Phyllanthus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Sauropus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Hibiscus meraukensis	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Grevillea mimosoides	---	---	---	---	---	---	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Dodonaea polyandra	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Rhynchosia minima	---	---	---	---	---	---	9.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Lepturus repens	---	---	---	---	7.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Ficus fraseri	---	---	---	---	2.4	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Ixora timorensis	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia hemignosta	---	---	---	---	---	---	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia simsii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Clerodendrum inerme	---	---	---	---	3.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Psychotria loniceroides	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Exocarpos latifolius	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Pittosporum	---	---	---	---	2.4	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Leucopogon yorkensis	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia calyculata	---	---	---	---	---	---	---	---	---	---	---	---	2.9	---	---	---	---	---	---	---	---	---	---	---
Gardenia scabrella	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Capparis	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Cryptocarya exfoliata	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Neoroepera banksii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Indigofera	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Melaleuca arcana	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Pandanus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Alyxia spicata	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Dalbergia densa var. australis	---	---	---	---	---	---	---	2.2	9.5	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia hylandii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Terminalia muelleri	---	---	---	---	---	---	---	8.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Siphonodon pendulus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4	---	---	---	---	---	---
Melaleuca stenostachya	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Erythrina vespertilio	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Sersalisia sericea	---	---	---	---	---	---	---	2.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Bursaria incana	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Canarium australianum	---	---	---	---	---	---	---	3.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Psychotria	---	---	---	---	---	3.3	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia confertiflora	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.8	---	---	---	---	---	---
Flueggea virosa subsp. *melanthesoides*	---	---	---	---	---	---	---	1.5	---	---	---	---	---	---	---	2	---	---	---	---	---	---	---	---
Caladenia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Acacia oraria	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Wrightia saligna	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Erythroxylum ellipticum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.2	---	---	---
Bridelia tomentosa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.5	---	---	---	---	---
Grevillea pteridifolia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.9	---	---	---
Hovea	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	6.2	---	---	---
Lomandra banksii	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.4	---	---	---
Croton arnhemicus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	4.2	---
Abrus precatorius	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.1	---
Buchanania arborescens	---	---	---	---	---	---	---	---	---	2.2	---	---	---	---	---	---	---	---	---	---	---	2.2	---
Manilkara kauki	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	2.5	---
Shonia tristigma	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	8.3	---	---	---
Psydrax	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.9	---	---	---
Callitris intratropica	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	6.2	---	---	---
Erythrophleum chlorostachys	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	3.1	---	---	---	---	---
Dischidia major	---	---	---	---	---	6.5	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Atalaya variifolia	---	---	---	---	---	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Chionanthus ramiflorus	---	5.3	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Brachychiton	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	1.5	---	---	---	---	---
Bursaria spinosa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Terminalia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Rubiaceae	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Syzygium suborbiculare	---	---	---	---	---	---	---	---	---	---	5.7	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia papuana	---	---	---	---	---	---	6.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Piliostigma malabaricum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Corymbia stockeri subsp. *stockeri*	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Planchonia careya	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	1.2	---	---	---	---	---

Ficus opposita	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	1.4	---	---	---	---	---	---
Denhamia oleaster	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Atalaya hemiglauca	---	---	---	---	---	---	9.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Grevillea glauca	---	---	---	---	---	---	---	---	---	---	---	---	8.2	---	---	---	---	---	---	---	---	---	---	---
Parinari nonda	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Lomandra	---	---	---	---	7.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Petalostigma pubescens	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Eucalyptus crebra	---	---	---	---	---	---	---	---	---	---	---	---	---	4.6	---	---	---	---	---	---	---	---	---	---
Xylomelum scottianum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Melaleuca nervosa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Drypetes deplanchei	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Santalum lanceolatum	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Coelospermum decipiens	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Alectryon tomentosus	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia disparrima subsp*. calidestris*	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Mallotus philippensis	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Urena lobata	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Sterculia quadrifida	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Lithomyrtus retusa	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Dodonaea malvacea	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Cayratia trifolia	---	---	---	---	3.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Eucalyptus platyphylla	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Platysace valida	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Grewia latifolia	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Antidesma ghaesembilla	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Carissa ovata	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

Synoptic table of groups from Fuzzy Noise Clustering with characterising species. Characterising species calculated on Fidelity with all groups standardised to equal size and *p <0.05*

Percentage synoptic table with fidelity (Phi coeff. C) (38 columns)

No. sites	1	1	1	1	1	1	1	1	1	1	1	2	2	2	2	3	2	3	4	4	2	1	6	4	5	4	6	7	8	11	11	1	1	1	1	1	1
Terminalia arenicola	19.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Eugenia reinwardtiana	19.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia leptostachya	19.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Dodonaea viscosa	17.1	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Premna serratifolia	13.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Petalostigma banksii	---	14.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	5.7	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Ectrosia	---	2.2	---	15.8	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Polycarpaea spirostylis	---	0.3	---	13.6	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Eriachne	---	2.7	---	12.2	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Acacia humifusa	---	---	---	---	14	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Imperata	---	---	---	---	---	27.9	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---	---

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Appendix 3.4 Clustering of sites by agglomerative hierarchical clustering (AHC) and fuzzy noise clustering (FNC)

Site Id	AHC Group	FNC Group	Site Name	Structural Formation	Community Context	Locality	Recorders
3591	r	F37	DIX 18	LOW	E.CULLENII, C.CLARKSONIANA OPEN WOODLAND-LOW OPEN WOODLAND; SOILS CYP 592	4.5 KM E OF THE COLEMAN RIVER ON THE EDWARD RIVER ROAD	John Neldner,John Clarkson
3592	r	F28	DIX 19	W	E.CULLENII, C.CLARKSONIANA WOODLAND	5 KM W OF THE PENINSULA DEVELOPMENTAL ROAD ON THE ROAD TO DIXIE	John Neldner
3596	w	F7	DIX 23	W	C. DALLACHIANA, M.VIRIDIFLORA, P.BANKSII WOODLAND	36.3 KM S OF DIXIE	John Neldner,John Clarkson
3597	l	F28	DIX 24	W	E.TETRODONTA WOODLAND; SOILS CYP 464	6.4 KM E OF KILLARNEY HOMESTEAD	John Neldner,John Clarkson
3759	u	F14	TEMP 2	DOHT	Acacia humifusa, Petalostigma pubescens dwarf open-heath on exposed headland	BOLT HEAD	John Neldner,John Clarkson
3786	r	F37	FEL 6	W	E.CULLENII, C.CLARKSONIANA SHRUBBY WOODLAND WITH GRANITE BOULDERS ON SURFACE	17.9 KM N OF MUSGRAVE, N OF BAMBOO STATION	John Neldner,John Clarkson
3787	r	F37	FEL 7	W	E.CULLENII, M.STENOSTACHYA WOODLAND	11.1 KM N OF THE BIG COLEMAN RIVER	John Neldner,John Clarkson
3788	l	F35	FEL 8	W	C.CLARKSONIANA, E.TETRODONTA, E.LEPTOPHLEBA WOODLAND WITH C.NESOPHILA (Original MU=92)	4.7 KM N OF DUCK HOLES CREEK ON THE PENINSULA DEVELOPMENTAL ROAD	John Neldner,John Clarkson
3792	o	F20	FEL 12	OW	E.CHLOROPHYLLA OPEN WOODLAND WITH E.LEPTOPHLEBA; SOILS CYP 572	12.5 KM FROM THE STEWART RIVER ON THE OLD ROAD	John Neldner
3797	l	F28	FEL 17	W	Eucalyptus tetrodonta, Erythrophleum chlorostachyus woodland	38.2 km north of the Coen airstip turnoff, 5.1 km south of the Archer River crossing	John Neldner,John Clarkson
3825	w	F34	LOC 3	OW	M.VIRIDIFLORA OPEN WOODLAND WITH EMERGENT E.TETRODONTA, C.CLARKSONIANA	34.9 KM ALONG THE PORTLAND ROADS ROAD FROM THE PENINSULA DEVELOPMENTAL ROAD	John Neldner,John Clarkson

3826	r	F37	LOC 4	W	E.CULLENII, C.CONFERTIFLORA WOODLAND	36.2 KM FROM THE PENINSULA DEVELOPMENTAL ROAD ON THE LOCKHART RIVER ROAD	John Neldner,John Clarkson
3827	r	F37	LOC 5	W	Woodland of Euc.cullenii on granite hills.	49.2 KM NE OF THE PENINSULA DEVELOPMENTAL ROAD ON THE LOCKHART RIVER ROAD	John Neldner
3831	m	M1	LOC 9	W	E.TETRODONTA, C.HYLANDII WOODLAND WITH M.VIRIDIFLORA	1.8 KM E OF BROWN CREEK	John Neldner,John Clarkson
3833	m	F27	LOC 11	LOF	A.BRASSII, N.FYRTIFOLIA LOW OPEN FOREST WITH EMERGENT E.TETRODONTA, E.NESOPHILA	1.4 KM E OF GARRAWAY CREEK	John Neldner,John Clarkson
3839	l	F38	LOC 17	W	E.TETRODONTA WOODLAND WITH C.CLARKSONIANA	QUINTEL BEACH, 200 M SE OF THE BARGE LANDING, 200 M FROM LOC 16 SITE	John Neldner,John Clarkson
3840	l	F28	LOC 18	W	E.TETRODONTA WOODLAND	SOUTH OF LOCKHART RIVER ON THE ROAD TO THE LANDING, 1 KM INLAND FROM THE BEACH	John Neldner,John Clarkson
3842	l	F38	LOC 20	W	E.TETRODONTA WOODLAND	3.7 KM S OF LOCKHART RIVER	John Neldner,John Clarkson
3844	m	F34	LOC 22	LW	M.VIRIDIFLORA, A.LITTORALIS, N.MYRTIFOLIA LOW WOODLAND EXPOSED TO WINDS	7.5 KM FROM THE OLD INTERSECTION ON THE ROAD TO PORTLAND ROADS	John Neldner,John Clarkson
3845	m	F27	LOC 23	LW	A.BRASSII, A.BRASSII, A.LITTORALIS LOW WOODLAND	8.8 KM FROM THE PORTLAND ROADS ROAD	John Neldner,John Clarkson
3846	p	F35	LOC 24	W	E.LEPTOPHLEBA WOODLAND	8.3 KM S FROM THE PASCOE RIVER	John Neldner,John Clarkson
3847	m	F36	LOC 25	OF	C.NESOPHILA, E.TETRODONTA OPEN FOREST-WOODLAND	12.2 KM N OF THE OLD TURNOFF TOWARDS PORTLAND ROADS	John Neldner,John Clarkson
3848	m	F36	LOC 26	OF	C.NESOPHILA, E.TETRODONTA WOODLAND	1.9 KM S OF PORTLAND ROADS	John Neldner,John Clarkson
3851	q	F24	LOC 29	W	E.BRASSIANA WOODLAND	0.1 KM FROM THE LOCKHART TO CHILI BEACH TURNOFF	John Neldner,John Clarkson

3859	m	F38	LOC 37	W	E.TETRODONTA WOODLAND WITH C.NESOPHILA, MELALEUCA SPP.	7.4 KM W OF THE OLD MISSION SITE, LOCKHART ABORIGINAL RESERVE	John Neldner,John Clarkson
3864	t	F31	LOC 42	OSC	L.PURPURASCENS TALL SHRUBLAND	2.4 KM E OF BROWN CREEK, 24.5 KM W OF THE OLD LOCKHART-PORTLAND ROADS TURNOFF	John Neldner,John Clarkson
3868	u	F18	OLI 1	OHT	A.LYSICEPHALA, J.THESIOIDES OPEN HEATH (Original MU=171)	8.5 KM N OF PORTLAND ROADS, ON THE ROAD TO WATTLE HILLS	John Neldner,John Clarkson
3869	h	F30	OLI 2	W	C.HYLANDII, W.LONGIVALVE WOODLAND	9.2 KM N OF PORTLAND ROADS, ON THE ROAD TO WATTLE HILLS	John Neldner,John Clarkson
3870	t	F31	OLI 3	TS	L.PURPURASCENS TALL SHRUBLAND	9.4 KM FROM THE ROAD JUNCTION, THE HILL TO THE E OF THE ROAD, 300 M FROM SITE 2	John Neldner,John Clarkson
3873	m	F36	OLI 6	W	C.NESOPHILA, E.TETRODONTA SHRUBBY WOODLAND (Original MU=82)	2.4 KM N OF WATTLE HILL	John Neldner,John Clarkson
3875	p	F35	OLI 8	W	E.LEPTOPHLEBA WOODLAND	APPROX. 30 KM NW OF WATTLE HILL	John Neldner,John Clarkson
3877	l	F38	OLI 10	W	E.TETRODONTA, C.CLARKSONIANA WOODLAND	1.8 KM W OF THE BEACH, N OF THE PASCOE RIVER MOUTH	John Neldner,John Clarkson
4098	f	F17	BAM 15	TG	ARISTIDA SP., SCHIZACHYRIUM FRAGILE TUSSOCK GRASSLAND; NEAR SOILS CYP 70	8.3 KM N OF THE TURNOFF FROM SOMERSET, 400 M E OF THE ROAD, MT BREMER	John Neldner,John Clarkson
4100	i	F33	BAM 17	W	C.HYLANDII, E.CULLENII WOODLAND; SOILS CYP 72	9.5 KM FROM THE TURNOFF ON THE MAIN ROAD VIA PUNSAND BAY	John Neldner,John Clarkson
4174	n	F9	WAK 29	LW	C. HYLANDII LOW WOODLAND; SOILS CYP 274	MID-SLOPE OF CAPE MELVILLE, IN THE NATIONAL PARK, 750 M FROM THE CREEK	John Neldner,John Clarkson
4175	b	F19	WAK 30	LOF	C. GILLIVRAEI, E. CREBRA LOW CLOSED FOREST; MAINLY DECIDUOUS IN DRY SEASON (Original MU=126)	MID-SLOPE OF THE NW FACE, CAPE MELVILLE NATIONAL PARK	John Neldner,John Clarkson
4176	q	F24	WAK 31	OF	E. CLARKSONIANA, E. BRASSIANA OPEN FOREST; SHELTERED BEHIND LARGE BOULDERS	MID-SLOPE OF NW FACE, CAPE MELVILLE, CAPE MELVILLE NATIONAL PARK	John Neldner,John Clarkson

4177	r	F37	WAK 32	OF	E. CULLENII, E. CLARKSONIANA OPEN FOREST; E. CLARKSONIANA ON ADJACENT FLAT	FOOTSLOPES OF CAPE MELVILLE, CAPE MELVILLE NATIONAL PARK	John Neldner,John Clarkson
4251	m	F36	HEL 41	OF	C.NESOPHILA, E.TETRODONTA SHRUBBY WOODLAND WITH BOULDERS COVERING SURFACE	CA 26 KM WSW OF LOCKHART RIVER TOWNSHIP, CA 7 KM SSE OF MOUNT BOWDEN	John Neldner,John Clarkson
4252	m	F27	HEL 42	LW	N.MYRTIFOLIA, A.BRASSII, L.SUAVELOENS LOW WOODLAND; MUCH ROCK ON PART OF PLOT	CA 26 KM WSW OF LOCKHART RIVER TOWNSHIP, CA 7 KM SSE OF MOUNT BOWDEN	John Neldner,John Clarkson
4253	t	F31	HEL 43	TS	L.PURPURASCENS TALL SHRUBLAND; MUCH ROCK PAVEMENT	CA 26 KM WSW OF LOCKHART RIVER TOWNSHIP, CA 7 KM SSE OF MOUNT BOWDEN	John Neldner,John Clarkson
4254	h	F5	HEL 44	LW	W.LONGIVALVE, M.VIRIDIFLORA, N.MYRTIFOLIA LOW WOODLAND (Original MU=148)	CA 30 KM WNW OF LOCKHART RIVER TOWNSHIP, CA 6 KM N OF MOUNT BOWDEN	John Neldner,John Clarkson
4255	t	F21	HEL 45	TS	Leptospermum purpurascens, Canthium sp. tall shrubland	CA 30 KM WNW OF LOCKHART RIVER TOWNSHIP, CA 6 KM N OF MOUNT BOWDEN	John Neldner,John Clarkson
4256	m	F27	HEL 46	LW	N.MYRTIFOLIA, A.BRASSII, A.LITTORALIS LOW WOODLAND WITH Melaleuca arcana	2.8 KM N OF GARRAWAY HILL, CA 25 KM WNW OF LOCKHART RIVER TOWNSHIP	John Neldner,John Clarkson
4257	u	F34	HEL 47	DS	Asteromyrtus lysicephala, Melaleuca viridiflora dwarf heath(Original MU=178)	2.9 KM N OF GARRAWAY HILL, CA 25 KM WNW OF LOCKHART RIVER TOWNSHIP	John Neldner,John Clarkson
4262	l	F38	HEL 52	W	E.TETRODONTA, C.STOCKERI SUBSP PENINSULARIS WOODLAND	1.6 KM SW OF PRATT TIN MINE, CA 17 KM ESE OF WOLVERTON	John Neldner,John Clarkson
4263	h	F30	HEL 53	LW	Welchidendron longivalve, Asteromyrtus lbrassii Low Woodland	CA 4 KM N OF PRATT TIN MINE, CA 17 KM ESE OF WOLVERTON	John Neldner,John Clarkson
4264	d	F29	HEL54A	OH	Sparse Herbland on Rock Pavement	CA 17 KM ESE OF WOLVERTON, CA 4 KM N OF PRATT TIN MINE	John Neldner
4267	p	F35	HEL 57	W	E.LEPTOPHLEBA, C.CLARKSONIANA WOODLAND	PASCOE RIVER, CA 29 KM WSW OF LOCKHART RIVER	John Neldner,John Clarkson
4268	m	F36	HEL 58	OF	Eucalyptus tetrodonta, Corymbia nesophila open-forest	PASCOE RIVER, CA 29 KM WSW OF LOCKHART RIVER	John Neldner,John Clarkson
4271	w	F34	HEL 61	LW	Melaleuca viridiflora, Welchidendron longivalve low woodland	CA 24 KM SSE OF THE OLIVE RIVER MOUTH, CA 3 KM S OF MOSQUITO POINT	John Neldner,John Clarkson

4272	h	F21	HEL 62	LOF	A.BRASSII LOW OPEN FOREST (Original MU=134)	CA 24 KM SSE OF THE OLIVE RIVER MOUTH, CA 3 KM S OF MOSQUITO POINT	John Neldner,John Clarkson
4273	r	F37	HEL 63	W	E.CULLENII WOODLAND WITH A.AULACOCARPA, D.POLYANDRA	CA 24 KM SSE OF THE OLIVE RIVER MOUTH, CA 3 KM S OF MOSQUITO POINT	John Neldner,John Clarkson
4275	c	F15	HEL 65	CTG	Imperata cylindrica, Mnesithea ROTTBOELLIOIDES CLOSED TUSSOCK GRASSLAND	CA 39 KM NNW OF LOCKHART RIVER, CA 12 KM W OF THE PASCOE RIVER MOUTH	John Neldner,John Clarkson
4276	p	F35	HEL 66	W	Eucalyptus leptophleba Woodland. Sheltered from winds	CA 39 KM NNW OF LOCKHART RIVER, CA 12 KM W OF THE PASCOE RIVER MOUTH	John Neldner,John Clarkson
4277	u	F6	HEL 67	OS	Allocasuarina littoralis, Choriceras tricorne open-heath	CA 28 KM NW OF LOCKHART RIVER, CA 29 KM W OF CAPE WEYMOUTH	John Neldner,John Clarkson
4278	t	F31	HEL 68	TS	Leptospermum purpurascens, Asteromyrtus brassii tall shrubland	CA 28 KM NW OF LOCKHART RIVER, CA 29 KM W OF CAPE WEYMOUTH	John Neldner,John Clarkson
4279	d	F29	HEL 69	OH	SPARSE HERBLAND ON ROCK PAVEMENT	CA 28 KM NW OF LOCKHART RIVER, CA 29 KM W OF CAPE WEYMOUTH	John Neldner,John Clarkson
4320	a	F16	JPS 3	LOF	L.SUAVEOLENS, E.CREBRA LOW OPEN FOREST WITH E.GLAUCA	10 KM E OF WAKOOKA, ALTANMOUI RANGE, 200M UPSLOPE FROM JPS 2	John Neldner
4341	m	F36	JPS 24	W	C.NESOPHILA, E.CREBRA WOODLAND	5.5 KM E OF SADDLE HILL ON AN UNNAMED SANDSTONE PLATEAU	John Neldner
4342	m	F36	JPS 25	W	E.NESOPHILA LOW WOODLAND	5.5 KM E OF SADDLE HILL ON AN UNNAMED SANDSTONE PLATEAU	John Neldner
4378	r	F37	ROK 33	LW	E.CULLENII, C.CLARKSONIANA LOW WOODLAND; SOILS 550	10.5 KM FROM OLD COEN ROAD ON THE ROAD TO PORT STEWART	John Neldner,John Clarkson
4382	l	F22	ROK 37	W	C.HYLANDII, E.TETRODONTA WOODLAND ON GRANITE	16.3 KM SW OF ATTACK CREEK ON THE COEN TO BIRTHDAY MOUNTAIN ROAD	John Neldner,John Clarkson
4383	r	F38	ROK 38	W	E.TETRODONTA, E.CULLENII, C.STOCKERI SUBSP PENINSULARIS WOODLAND; SOILS 555	9.9 KM FROM THE PENINSULA DEVELOPMENTAL ROAD, E OF THE ARCHER RIVER CROSSING	John Neldner,John Clarkson
4384	n	F25	ROK 39	LOW	C.DISJUNCTA LOW OPEN WOODLAND	16.2 KM FROM THE PENINSULA DEVELOPMENTAL ROAD, E OF THE ARCHER RIVER CROSSING	John Neldner,John Clarkson

4385	r	F37	ROK 40	W	E.CULLENII, C.CLARKSONIANA, E.LEPTOPHLEBA WOODLAND; SOILS 556	16.2 KM FROM THE PENINSULA DEVELOPMENTAL ROAD, E OF THE ARCHER RIVER CROSSING	John Neldner,John Clarkson
4386	p	F35	ROK 41	OW	E.LEPTOPHLEBA, C.CLARKSONIANA OPEN WOODLAND	18.4 KM E OF ARCHER RIVER CROSSING ON THE PENINSULA DEVELOPMENTAL ROAD	John Neldner,John Clarkson
4387	p	F4	ROK 42	W	E.LEPTOPHLEBA, E.PLATYPHYLLA, C.CLARKSONIANA WOODLAND	19.7 KM E OF ARCHER RIVER CROSSING ON THE PENINSULA DEVELOPMENTAL ROAD	John Neldner,John Clarkson
4389	r	F37	ROK 44	LW	E.CULLENII, M.VIRIDIFLORA LOW WOODLAND	50.3 KM NE OF ARCHER RIVER CROSSING, PDR, WOLVERTON ROAD VIA COOK TIN MINE	John Neldner,John Clarkson
4390	n	F25	ROK 45	W	Corymbia disjuncta, Erythrophleum chlorostachyus grassy woodland; SOILS CYP 759	2.2 KM W OF ORCHID CREEK HOMESTEAD	John Neldner,John Clarkson
4416	r	F38	POM 1	W	E.TETRODONTA, C.CLARKSONIANA, E.CULLENII WOODLAND; SOILS 593	15.2 KM W OF MUSGRAVE	John Neldner,John Clarkson
4471	s	F26	POM 56	LOW	M.CITROLENS, M.FOLIOLOSA, E.CULLENII LOW OPEN WOODLAND	1.5 KM E OF THE HOLROYD RIVER CROSSING ON THE HOLROYD ROAD	John Neldner,John Clarkson
4472	o	F20	POM 57	OW	E.CHLOROPHYLLA, E.CONFERTIFLORA, E.CULLENII OPEN WOODLAND; SOILS 649	10.7 KM E OF THE HOLROYD RIVER CROSSING ON THE HOLROYD TO YARRADEN ROAD	John Neldner,John Clarkson
4475	p	F35	POM 60	W	E.LEPTOPHLEBA SHRUBBY WOODLAND; SOILS 652	12 KM NW OF COEN	John Neldner
4476	q	F24	POM 61	OF	E.CLARKSONIANA, E.BRASSIANA OPEN FOREST; SOILS 653	12 KM NW OF COEN	John Neldner,John Clarkson
4477	r	F37	POM 62	W	E.CREBRA, E.CLARKSONIANA WOODLAND; SOILS 654	10 KM NNW OF COEN	John Neldner,John Clarkson
4478	b	F8	POM 63	LW	CO.GILLIVRAEI, CA.AUSTRALIANUM DECIDUOUS LOW WOODLAND; SOILS 655	9 KM NNW OF COEN	John Neldner,John Clarkson
4489	m	F22	HRN 1	LW	C.HYLANDII, W.LONGIVALVE LOW WOODLAND WITH C.NESOPHILA (Original MU=74)	1.5 KM E OF WASAGA, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4491	i	F33	HRN 3	W	C.HYLANDII WOODLAND WITH W.LONGIVALVE	BEHIND THE UNIVERSITY CENTRE, 5 KM E OF WASAGA, HORN ISLAND, TORRES STRAIT	John Neldner

4493	h	F30	HRN 5	W	E.PLATYPHYLLA, W.LONGIVALVE WOODLAND	3 KM SW OF KING POINT, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4494	i	F33	HRN 6	OF	Corymbia stockeri subsp. peninsularis open-forest with C. tessellaris	2.5 KM S OF THE AIRSTRIP, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4496	h	F2	HRN 8	W	E.CULLENII, W.LONGIVALVE, A.BRASSII WOODLAND	NEAR THE QUARRY, 3 KM S OF THE AIRSTRIP, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4498	e	F32	HRN 10	TG	S.PLUMOSUM, H.TRITICEUS TUSSOCK GRASSLAND WITH DWARF SHRUBS (Original MU=177)	KING POINT, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4499	k	F23	HRN 11	S	E.TESSELLARIS, J.THESIOIDES SHRUBLAND; WIND & FIRE MAINTAINED	0.3 KM FROM KING POINT, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4501	h	F30	HRN 13	LOF	W.LONGIVALVE LOW OPEN FOREST (Original MU=124)	1 KM FROM CABLE BAY, HORN ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4502	i	F33	HRN 14	OF	Corymbia stockeri subsp. stockeri open-forest	1 KM W OF MURALUG, PRINCE OF WALES ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4512	h	F30	HRN 24	LOF	W.LONGIVALVE, T.MUELLERI LOW CLOSED FOREST - LOW CLOSED SCRUB (Original MU=124)	2 KM E OF BRAMPFIELD HEAD, PRINCE OF WALES ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4521	k	F23	HRN 33	OF	C.TESSELLARIS, W.LONGIVALVE OPEN FOREST (Original MU=44)	1.5 KM S OF BANKS PEAK, MOA ISLAND, TORRES STRAIT	John Neldner,John Clarkson
4523	m	F36	HRN 35	OF	E.NESOPHILA OPEN FOREST (Original MU=83)	GREEN ANT HILL, HORN ISLAND, TORRES STRAIT	John Neldner
4524	h	F30	HRN 36	LW	W.LONGIVALVE, C.GILLIVRAEI LOW WOODLAND (Original MU=126)	GREEN ANT HILL, HORN ISLAND, TORRES STRAIT	John Neldner
4641	v	F10	ISL 41	OSC	C.GILLIVRAEI, T.ARENICOLA OPEN SCRUB	RESTORATION ISLAND	
4662	e	F32	ISL 62	CTG	H.TRITICEUS CLOSED TUSSOCK GRASSLAND	SIR CHARLES HARDY ISLANDS, SOUTH ISLAND	John Clarkson
4761	g	F11	RUT 21	OTG	SCHIZACHYRIUM SP., RHYNCHOSPORA SP. OPEN TUSSOCK GRASSLAND WITH FEW LOW TREES	3 KM E OF THE LUKIN RIVER AND BAMBOO CREEK JUNCTION	John Neldner
4763	g	F13	RUT 23	TG	Ectrosia sp., Eriachne sp., Schizachyrium sp. tussock grassland	22 KM NNW OF GLEN GARLAND, ON SWAMP CREEK	John Neldner

4815	e	F32	GBR 30	OTG	H.TRIICEUS OPEN TUSSOCK GRASSLAND ON SAND	SIR CHARLES HARDY ISLAND, NW ISLAND IN THE GROUP, GREAT BARRIER REEF	
4818	e	F32	GBR 33	CTG	H.TRITICEUS CLOSED GRASSLAND	SIR CHARLES HARDY ISLAND, GREAT BARRIER REEF	
24094	l	F38	WAL037_72	LW	Eucalyptus tetrodonta low woodland with a shrub layer of juvenile trees and a sparse ground layer on coarse sand on a plain.	20 km S of Strathleven HS on the road to Gamboola.	Gary Wilson
49755	s	F26	HAN09	LOW	Open woodland of Melaleuca citrolens with scattered Terminalia platyptera on low granite hills.	11.5 kms along Pinnacle Rd from Kimba rd and 200m to west, CYP. Transect @ 215 from 0-50m.	John Neldner,Eda Addicott,Mark Newton
49757	s	F26	HAN11	LOW	Low open woodland of Melaleuca citrolens and Petalostigma banksii on slopes of low granite rises.	12.8kms north of King Junction on Pinnacles rd and 150m west of rd., CYP	John Neldner,Eda Addicott,Mark Newton
49758	l	F38	HAN12	W	Woodland of Eucalyptus tetrodonta and Corymbia stockeri subsp. peninsularis on broad plain.	3kms along track heading north off Pinancles rd 10kms north of King Junction house, CYP.Transect @ 125 from 0-50m.	John Neldner,Eda Addicott,Mark Newton
49780	l	F38	STRATH02	OW	Open woodland of Eucalyptus tetrodonta with Corymbia clarksoniana on low rises.	13.5 kms south of Palmer River Crossing on the Gamboola Rd and 100m west of rd. Transect @ 300 from 0-50m.	John Neldner,Eda Addicott,Mark Newton
52914	l	F38	COE1	W	Woodland of Eucalyptus tetrodonta and Corymbia stockeri subsp. peninsularis on the slopes of a low granite hill.	15kms south of the Pascoe River crossing on the Lockhart River road, CYP.	John Neldner,Eda Addicott,Mark Newton
52935	e	F32	WEY2	TG	Grassland dominated by Hetropogon triticeus on exposed granite headland.	Approx. 500m south east of Portlands Roads on exposed granite headland.	John Neldner,Eda Addicott,Mark Newton
52936	j	F12	WEY3	LOF	Low open forest of Acacia polystachya and Dodonea polyandra on steep slopes of a granite headland.	Approx 400m south east of Portlands roads on headland, CYP.	John Neldner,Eda Addicott,Mark Newton
52937	w	F34	WEY1	LW	Low woodland of Melaleuca viridiflora with scattered Eucalyptus leptophleba on low granite hills.	1.3kms along private road off turnoff to Chili Beach, CYP.	John Neldner,Eda Addicott,Mark Newton

52938	w	F34	COE3	W	Woodland of Melaleuca viridflora and Corymbia clarksoniana on rolling low granite footslopes.	6kms north of the Wenlock River crossing on the road to Lockhart River and apprx. 50 east of the road, CYP.	John Neldner,Eda Addicott,Mark Newton
52954	w	F3	WEY7	LOW	Low open woodland of Melaleuca viridiflora, Lophostemon suaveolens and Acacia brassii in steep slopes of a low granite hill.	5.5kms east of Garraway Creek on road to Lockhart River and 250m up the hill to south of rd.	John Neldner,Eda Addicott,Mark Newton

Appendix 3.5 Workshop minutes

Minutes of workshop on the numerical classification of plant communities forming REs in CYP Bioregion: Land Zone 12 as a case study

Tuesday Dec 2nd, 11am – Friday Dec 5th noon, 2014.

Attendees:

Tuesday 2nd Dec: Eda Addicott (EA), Mark Newton (MN), Jeanette Kemp (JK), John Neldner (JN), Don Butler (DB), Bruce Wannan (BW), Darren Crayn (DC), Sue Laurance (SL).

Wednesday 3rd & Friday 5th Dec: Eda Addicott, Mark Newton, John Neldner, Don Butler.

Summary of Final Outcomes from Workshop:

Sample adequacy

Will need to use date as a filter for ground layer rather than any attribute in corveg.

General agreement on formalised 2 tiered approach with respect to sites for numerical analysis and then observational sites for areas we don't have sites in.

Abundance measure

Agreement to use cover data in analysis

Rainforest and wet sclerophyll communities

Use Neldner and Lynch 'not' definition to choose whether a site is a rainforest.

Test using P/A with height in data matrix for rainforest classification

Include wet sclerophyll sites in rainforest and non-rainforest classification analysis

Species to include in classification analysis

Use species that contribute >1% to Total Foliage Cover at a site

Incorporating height and strata

Retain all data in a layer regardless of % total foliage cover in the strata.

Decision to incorporate structure by using height in metres after looking at clusters one by one.

Internal evaluators

Use available evaluators to assist with choosing optimal clusters rather than sticking to one particular evaluator. In the expert panel review of clusters all these evaluators will be used.

Testing classification techniques

Use UPGMA. No need to test other techniques.

Incorporating ground layer

Generally, analysing ground layer separately and overlaying woody clusters to look for possible divisions should work for a wider area.

Including environmental variables

Landscape is a secondary consideration in defining groups. Floristic composition is the primary consideration.

Environmental variables and landform are used to understand the vegetation communities, not to define them.

‘Landscape’ criteria to be used as a guideline and not included in quantitative analysis. Concerns that landscape criteria have not been consistently recorded to be reliably used in a numerical analysis, but can be consulted later as potential qualifiers or explainers.

Process for including numerical analysis results with expert review

Numerical analysis is NOT about slavishly re-creating the classification process. It’s about augmenting the expert-based methodology.

The floristic composition is the primary consideration in the delineation and recognition of new vegetation communities within a land zone. Landform, landscape and geological formations are secondary and only used to better understand the floristic patterns.

The main aim of our descriptions are to communicate the core idea of the RE or vegetation community along with the variability shown by our sites.

Guiding principle is not to lose information.

Analyse sites with a woody and ground layer EDL together, but exclude ground layer data from sites with a woody EDL.

Produce clusters using Bray-Curtis similarity coefficient and UPGMA clustering. Produce optimal clusters from an array of evaluators. These become ‘proposed communities’.

General process is to investigate ‘proposed communities’ resulting from the evaluators. Consider any differences from subjective ‘proposed communities’ and allocation of sites to that ‘proposed community’ in light of expert knowledge, observational sites and other info not incorporated in site data.

Where the sites allocated subjectively to an RE are split among clusters this is a trigger for closer investigation.

Further investigation includes (but is not limited to) looking at original site sheet, observational sites, spatial distribution of map unit and possibly duplicating site in cluster analysis to see the effect it has on the clusters.

If still don’t agree with cluster outcome, site to be marked as ‘not-representative’, but variation put in as a comment in REDD description along with the corveg id of site.

Map unit to be retained, but incorporated into the appropriate vegetation community. This mapped variability (i.e. geographic distribution) also to be included in comments in REDD description.

Future work

This method for including ground layer seems to work but more testing needed to check against other possible methods.

Re-do lz 12 sites for final groupings using modifications suggested by workshop

exclude RF sites (as per the ‘not’ definition)

Change the ‘included species’ to 1% for all formations

Keep in all data for all layers (i.e. don’t delete data in S, T2 T3 if I<10% TFC)

Check the clusters resulting from our expert review and see if they’re adding value to classification.

Test suggested other methods for inclusion of ground layer.

Do whole of bioregion analysis with all sites, for woody’s and for ground layer

Question – Do the clusters from this support the broad veg groups?
What is the concordance of the clusters from this analysis and the subjective broad veg groups?

Does the ground layer cluster analysis support the land zones?

Look at land zone 5 cluster analysis

Re-do land zone 1 cluster analysis with new methods

Collate and document our subjective methodology for determining site adequacy.

Background and theoretical framework for classification

On assumptions...

JN: Sites tied with classification for mapping. Placement and collection of site data driven by mapping at 1:100K.

JK: Sites driven by mapping, not to describe ecological communities. Not driven by random sampling strategies.

DB: sites are also then used as a biodiversity surrogate, for example as benchmarks for bio-condition work.

Sampling adequacy

JN & BW have been involved with some trials with real-time attempts at collecting data from randomly placed sites but entailed a lot of wasted time, such as access, sites being over the edge of cliffs etc.

JN: Land zones 12 (&11 and 10) sample adequacy of sites is lower (harder country) than LZ 3, 5 because access is harder and roads are on LZ 5, 3, etc.

JN: Important to reach taxa sampling asymptotes by doing a number of sites to enable good species sampling within a pattern.

General discussion was that need a balance between time doing a site comprehensively versus the number of sites done was important.

BW: comment that as sampling adequacy (SA) derived from soil, and that as veg more complex the SA may not be sufficient.

EA: Landscape complexity is aligned with photo-pattern complexity and that if you're sampling the pattern complexity adequately you should be sampling the landscape complexity adequately.

DB: argument that soils are just as complex but some characters harder to see.

SL: In numerical classification sites are based on fine scale so limitations occur in scaling up to map scales.

JK: Easier in west to use photo-patterns to define communities but harder in wetter areas (i.e. WET, CQC and particularly rainforests).

JN: So far this analysis is taxa based (no landform, structure etc.) so is limited for RE descriptions as such. (EA structure is incorporated). *See final outcomes resulting from discussions through rest of workshop.*

General discussion on how to, or whether to, use ground layer differences to distinguish communities / REs on ground layer as it is specified in the methodology. JK suggested using initial woody clusters and then running analysis on these.

JN: sees numerical analysis not as an end point, but as a start point for discussion by the expert panel (technical review panel).

SL: Need to consider numerical classification with respect to vegetation types (i.e. savannas, grasslands, rainforests) and use them differently. Strongly suggested not including rainforests in with savannas and grasslands.

EA: This is OK but then method has to be consistent across state

DB: logical to use new sites to revisit RE classification at later date after a number of extra sites are done.

SL: need to target areas that are under sampled for data collection.

VN: seasonal issues with ground layer with respect to analysis. Suggests in sites done after June or July then ground layer data will be limited.

DB: His view on using numerical analysis is to check current mapping and pick up some things that need closer scrutiny. And then using other related data, i.e. obs. sites, spatial data to re-check.

EA: Ian Oliver says problems with on the fly subjective mapping. This is reason for needing consistent strict evaluation methods to remove subjectivity.

JK: Described differentiation between comprehensive species list at time of survey as opposed to comprehensive species list in site across the whole year. For the sites that Jeanette and Rosemary collected , a "full species list" just meant that all species present were included, no matter what time of year the sampling was done. In Wet Tropics and CQC they generally only sampled when the vegetation was good (which might have varied from year to year according to rainfall). Also in some veg types (eg wet tropics wetlands) the best time of year is much later when the wetland is drying and sedges are fruiting. Her perception was that other mappers were using "full species list" in the same way as her (i.e. calling it a full species list no matter what time of year). There might be a need to filter ground layer data on the time of year, but suspect not many will need to be removed for Wet Tropics and CQC.

Outcome:

Will need to use date as a filter for ground layer rather than any attribute in corveg.

Post workshop discussion: The date to be used as a flag for manually checking whether ground layer data is useable.

General agreement on 2 tiered approach in that we use numerical analysis to define proposed communities when we have detailed site data and then observational sites for areas we don't have sites in.

2 tiered approach also in that numerical analysis used initially for defining proposed plant communities – this is then reviewed by expert panel.

Which abundance measure to use?

Some discussion about abundance measure to use in RF - discussed later.

SL: her concern with using cover only is that it may not recognise the difference in age structure between two sites with the same species. Cover doesn't allow for structural differences (which could be a result of events like fire).

Some discussion about condition and disturbance in communities and how that is addressed in mapping.

Outcome:

Agreement to use cover data in analysis

Sites with dense crown cover & RF classification

Some discussion about the hypotheses presented – but pointed out that these were just hypotheses to test

SL: Don't use 2 indices to graph against each other as they can be auto-correlated – use raw data (i.e. species number)

Do we include floristic classification with the existing structural classification?

Consensus .yes we include floristics as well as structural classification.

If so – do we use P/A?

JK: Need to make sure collect basal area as it may be useful in future.

DB: P/A picks up regional difference in rainforest types.

Do we include wet sclerophyll sites with RF?

Use wet sclerophyll sites in both the P/A analysis for RF as well as the quantitative analysis for woodlands etc.

How do we define them?

EA: do we use dominants in description ? Suggests using a species list instead of dominants.

JN: Often sites are inadequately sampled and this complicates using them in a P/A analysis. So still need expert revision in outcomes of analysis. *(Also need to filter sites to ensure good sites in analysis)*

Does using P/A = change in data collection methods?

No, not for dense, diverse wet sclerophyll – and don't use my suggested cut-offs for analysis or data collection.

For wet sclerophyll as well as RF?

SL: questions idea of T2 removal. Consensus is that T2 defines community in wet sclerophyll as well as rainforest and need to keep it in.

Other considerations:

No resolution on how to filter rainforest sites for inclusion in numerical analysis.

Implications for bio-condition methodology?

JN: maybe value in adding some life form info.

What to do next?

JK: Questioned whether there was enough sites for analysis? **EA:** Yes for sampled patterns.

JN: issues with area of sampling (i.e. Webb & Tracey over large transects). Led to discussions about consistency of data.

JN: and other general discussion pointing out that floristics and site size will be extremely variable even for P/A data in rainforests. Some sites done by wandering as far as you could until stopped getting new species, other may be in a defined sample area, and also very dependent on observers identification skills.

SL: issues with time spent at site and experience of botanist, as per above.

SL: suggests removing rare species from matrix to see if strong patterns exist *(which will help overcome sampling method differences)*.

JK: using basal area would be good but EA says not consistent across state.

About using cover...

JK: possible suggestion to do cover but shorter transects for different canopy layers – i.e. nested transects.

EA: just do b.a. for dense wet sclerophyll sites as per criteria from research? (**JN** says no..still do covers).

Outcome:

Incorporate floristic classification in with structural classification.

Discussions by end of workshop resulted in idea to test using height with P/A to add structure into rainforest classification.

Include 'dense diverse woodlands' in with P/A classification and with quantitative classifications, but don't change data collection methods for these.

No resolution about how to deal with inconsistencies of site data / collection methods / site size with respect to using P/A in analysis, but will follow up SL suggestion.

Definition of rainforests: by the end of the workshop was to use the 'not' definition to define sites as RF. This is given in Lynch & Neldner paper in Australian Journal of Botany. Will be sites identified by this that will use different data collection method. All others stick to current corveg methodology.

When is a species included?

JN: Check time of year (consequences for ground layer species) of site collection. For ground layer don't use sites after July.

Some discussion about what to use in a cluster (e.g. *Galactia*) as a characterising species.

DC: Suggests that we should look at the species significantly associated with a cluster without prejudice about what is relevant. Take a step back...maybe this is important.

EA: has concern re ID but other (SL, DC) say in this instance ignore ID problems for now (later issue).

JN: Have you looked at sites in clusters to see how they compare with original RE allocations. (EA we will look at this for rest of workshop).

Discussion ensued about not using ground layer and risk of not picking up differences between 2 similar woodlands. (i.e. grass vs *Triodia*). EA asking for suggestions how to over comes this. JK suggest first pass for trees, then second pass for ground layer.

JK: not comfortable about subjectivity of choosing "important" species to establish ratios. What is a "useful" species. Why did EA remove some species as not useful? Could there be a more objective way to reduce the list? (SL suggest removing singletons (EA this means you risk losing valid proposed communities)) etc. OR clarify subjective process to satisfaction of peers.

DB: Why not just leave them all there?

EA: There were criteria to decide each exclusion/inclusion of 'useful' species.

[Criteria for 'not-useful' – are species responding to environmental variables at a micro-habitat scale rather than mapping scale? Are species ubiquitous and present in a number of communities, but not well represented because of sampling structure?]

JN: How different is this outcome [of the final list of species used] from current (say) EIU tech descriptions generated from corveg? His ideal is a LUCID key to determine RE's. EA argues this is the start of going toward a key. **BW:** let's take this through and compare to current subjective descriptions.

EA: what percent cut-off to use?

DB: suggest >1% across all formations as this get rid of very low covers and is still similar to 5% and 10%.

JN: Has some issues with these results being just on LZ12. From BVG perspective maybe look at all (say) heathlands on CYP. Suggests this needs further testing across rest of Cape.

JN: Better to keep more so use 1%.

SL: increased std dev in woodlands uniqueness is probably due to increased no of sites in woodlands.

Outcome:

Decision: across all structural formations use species that contribute > 1% to Total Foliage Cover at a site in analysis.

Incorporating height and strata.

SL: raised issue of problems with including height in community description possibly complicating the number of clusters.

JN: in some cases height is legitimate – as an example compared wind-swept vs sheltered *C. tessellaris* on Torres St. islands. Introduced idea of weighting to each layer.

General discussion about removal of ground layer. May be able to include if weightings used.

SL: Queried what clusters like without using height? EA: These included in results.

Deleting data in sites for layers where TFC is <10%

EA: leave it out or leave it in?

DB: may also create artificial differences (i.e. one layer 11% kept in and another close layer 9% removed).

Adding in the ground layers to analysis

DB: suggest separating on structural formation and then look within the groupings (i.e. *E. tetrodonta* woodlands).

Discussion ensued about not using ground layer and risk of not picking up differences between 2 similar woodlands. (i.e. tussock grass vs *Triodia*). JK suggest first pass for trees, then second pass for ground layer.

BW: May be should use BA's to check results against an existing data set. (i.e. species importance value vs basal area).

Outcome:

Retain all data in a layer regardless of % total foliage cover in the strata.

Height is o.k. to use as a measure of structure.

After considering diagrams etc. best to use H(m) as measure for incorporating height.

Test using H(m) as a way of including ground layer.

Choosing internal evaluators

SL: Important to say that one fits RE's better, but then validate it by saying why. Justify use of a specific evaluation? What are others doing that doesn't fit RE concept?

DB: happy for me to explore using the Lyons modelling approach to assessing classification solutions.

SL & DB: Follow up assessment required to compare ISA sites with the subjective RE assignment. Look at the range of different outcomes and assess why they are different or is there a need for more data. Be prepared to look at various evaluators (geometric, non-geometric, modelling) and assess the difference against current mapping/ look at with expert panel.

Outcome:

Use available evaluators to assist with choosing optimal clusters. In an expert panel review of clusters all these evaluators will be used.

General discussion of work so far

Does anyone think I should be testing against existing RE allocations?

Consensus: Not literally.

Should we re-sort individual sites by hand (when we don't agree with numerical analysis?)

DB: Not keen on changing RE based solely on analysis. Lengthy discussion about relevance of numerical classification to current REs.

DB, JN: Also talked about wanting to compare similar broad veg groups across different land zones and come up with a dissimilarity matrix for all REs.

DB: Need to incorporate the ground layer (possibly a 2 step process).

JK: Don't change the RE allocation of a site in corveg based on the analysis.

Assessment of 'proposed communities' – look at individual communities (*spatial distribution, technical descriptions, photos etc.)*

See separate minutes for individual decisions on sites and clusters.

Decision to use H(m) as the method for weighting strata, after assessment of individual clusters. Intuitively makes most sense after looking at structure diagrams etc. and the 'proposed communities' make sense to experts.

Suggest using the species contribution to Total Foliage Cover results as a rule of thumb. But use >1% for cut off for analysis.

General process is to investigate results from an array of evaluators and consider any differences in light of expert knowledge, observational sites and other info not incorporated in site data.

Welchiodendron / A. brassi cluster

Considered site 4272 and the fact that while it's clearly *Acacia brassi* dominated it is put into a *Welchiodendron* dominated cluster.

On closer inspection the *A. brassi* sites are separated at a lower cluster level – but not picked up by any of the evaluators as being a significant cluster. As part of the process of subjective expert review decide these divisions will be kept as an 'a' and 'b' as they occur in different geographical areas as well as being different associations.

On much closer inspection of site 4272 it was seen to be resulting from a fire and was not a climax community. Decide to put this as 'not-representative'.

Need to search 3D Torres vegetation units for *Acacia brassi*. Check whether they've identified *Acacia brassi* and *A. polystachia* or whether they may not have distinguished between the two and referred to everything as *A. polystachia.*

Sites 4416 , 4383 – clustered with *E. culleni*, but dominated by *E. tetrodonta*

Result of extra work showed that site 4383 is eco-tonal. Investigated sites, mapping, observational sites and the variation in clustering caused by duplicating the sites.

Decision that map unit 99 (which is where 4416 & 4383 were allocated) is an eco-tonal unit and really a variant of the individual *E. tetrodonta* and *E. culleni* units mapped in the area.

However, this area where *E. tetrodonta* and *E. culleni* occur together has been mapped and identified. This to be included as a comment in the REDD description.

Outcome

The main aim of our descriptions are to communicate the core idea or 'node' of the RE or vegetation community along with the variability shown by our sites.

Guiding principle is to not lose any information. So in situation where sites are allocated to a cluster that we still don't agree with after investigating site data etc. as per above, the process becomes:

Keep the map unit, but lump the map unit to the vegetation community. There may be a many to one relationship of map units to vegetation community.

Sites will be marked as 'not-representative' in corveg and not used in statistical analysis, technical descriptions or as diagnostic species etc. Add a comment to Corveg site of why not representative, e.g. ecotonal area most closely related to RE3.x.x

A comment about the variability is added to the REDD description along with the corveg id of the site showing the variability.

Which classification techniques to test?

Looked at preliminary results of fuzzy clustering (non-hierarchical partitioning), and recognised it would need a lot of work to adapt for our purpose. But, it did highlighted sites that expert panel queried.

General consensus that no point testing TWINSPAN.

UPGMA is giving good results in terms of making sensible groups suited for our classification purpose and is well supported in the literature. Literature also saying that flexible-beta clustering is often best suited to a variety of vegetation types and also that it is very close to UPGMA in classification effectiveness and groups produced.

Outcome

Decision to use UPGMA across rest of the state and no need to test other methods.

Eda would like to test other methods on land zone 12 for my own reassurance.

Incorporating ground layer

Ran cluster analysis on ground layer only, then overlayed this with clusters from woody analysis. Overstorey and understorey not relating to each other (put this comment into Masters).

At a gross level can separate ground layer into sites dominated by non-grasses, perennial grasses and annual grasses.

JN: concerned about the ground layer being confounded by the distinctive identifiable species being the main ones that are recorded and others not so easily identifiable being missed. General thought that this should be O.K. for distinguishing differences at an RE level.

Questioned whether there was a ground layer split evident between *E. tetrodonta* cluster and *C. nesophila* cluster. No evidence for this.

At a gross level can separate *E. tetrodonta* into a group with perennial grasses and a group with annual grasses.

Question: Can this split be seen in *E. tetrodonta* across the whole Cape? Does this reflect land zone splits?

Following on from this – include all sites across whole bioregion as an initial analysis to see what the major groups are at a bioregional level. Do this for woody data and for ground layer data. Are the BVG's supported by this analysis?

Outcomes:

Species aggregation is very important in ground layer. Need to make sure keep distinctive, recognisable taxa if possible (e.g. don't aggregate *Eriachne pallescens* into *Eriachne* sp.)

Running ground layer analysis separately and overlaying woody clusters looks like it's a workable method.

Eda would like to test other methods against this before settling on it.

Environmental variables

The 'landform element' and 'erosional pattern' on our site sheets aren't consistently well enough recorded across the state to use in analysis.

Consider deleting 'landform situation' from corveg site sheet. Not substantiated in any literature and is probably a relic from HERBRECS.

Environmental variables (e.g. landform, landscape, geological substrate) are used to look for correlations to increase understanding of the patterns and vegetation communities not as a criterion for defining vegetation communities.

Better to use the standard environmental variables for correlation such as precipitation, temperature etc. rather than the landform information recorded on site sheets.

Don't include the 'Landscape' criteria in the quantitative analysis.

Post workshop discussion: JK - "I think land zone should still be used to help define REs. (See previous comment) as we don't have enough data to do it entirely on floristics. Also REs are not just about plants, but also about animals and landscape processes."

Floristics are the primary division within a land zone. Floristic 'clusters' are not further split by landscape. If the environment is a real driver then this should be reflected in the floristic composition.

Post workshop discussion: JK disagrees with this – "we don't have enough data and we don't have fauna data. We are talking about ecosystems, not floristic associations".

Outcomes:

Environmental variables and landform are used to understand the vegetation communities, not to define them.

Post workshop discussion: JK disagrees with this outcome.

'Landscape' criteria to be used as a guideline and not included in quantitative analysis.

Floristic composition is the primary consideration in defining groups, with landscape as a secondary consideration.

General Topics

Allocation of new sites:

When get enough new sites to warrant a revision then re-do analysis. Generally a re-run of the analysis will be determined by external factors such as perceived need or accumulation of new sites.

Because the numerical analysis sits beside the subjective methodology new sites will be allocated subjectively.

The ultimate goal is to develop the equivalent of a LUCID key to regional ecosystems.

Edits required to Herb. Queensland Methodology

PFC in the RE criteria to be changed to crown cover to make it consistent with the rest of our outputs.

Corveg site types to be changed (when possible) from 'reconnaissance, detailed, observational' to match the standard 'quaternary, tertiary, secondary' nomenclature that is more widely used and understood.

Appendix 5: heading change ‘Criteria’ to ‘Guidelines’.

Add in ‘not’ in the last sentence of the structural criteria of ‘New regional ecosystem’.

Add ‘sub-canopy’ to criteria which talks about layers and delete ‘with at least 10% projective foliage cover’.

Italicise species names in this paragraph also.

JN and **EA** to edit the methodology. **JN** to find the master version of the methodology ‘working copy’.

Workshop closed: Friday 5th December, 12 noon.

Appendix 4

Appendix 4.1: Descriptive-framework for quantitatively derived vegetation communities on land zone 5 and 12 in Cape York Peninsula bioregion

I have retained the RE numbering system in this appendix. Land zone 5 = Tertiary landscape, land zone 12 = igneous landscape. 'Frequently occurring spp.' are produced using the technical-description routine in the Queensland government database 'CORVEG'. Frequency = % of total sites in which a species occurs; average %cover = mean of all cover values >0 for each species. Formatting is as outputted from this routine, with % after the frequency and no percent symbol associated with average %cover. For example, *Corymbia stockeri* (93%, 3) means this species occurs in 93% of sites and when it occurs has an average cover of 3%. Statistically **highly** associated species are those with a phi-coefficient of association >6 with a community. Statistically **significantly** associated are those significantly associated with a community ($p<0.05$) using Fisher's exact test. A species may be highly associated with a community but not significantly associated if it also has a strong association with another community. For example, *Dodonaea polyandra* is highly associated with RE 3.5.5, but not significantly as it is also highly associated with RE 3.5.42. Communities recognised using qualitative techniques are not included. *NA* = numerical analysis.

Version 10 RE *(number of sites used in NA)*	**Short label description**	**Frequently occurring spp. (% frequency, average %cover)** *(Species not occurring in the canopy layer are labelled with their layer, E = emergent, T2 = Second tree layer, S1 = shrub layer)*	**Statistically significantly associated spp.** *(p<0.05) (phi association coefficient)*	**Statistically highly associated spp.** *(phi association coefficient)*	**Occasional spp with high average %cover (% frequency, average %cover)** *(Species not occurring in the canopy layer are labelled with their layer, E = emergent, T2 = Second tree layer, S1 = shrub layer)*	**Additional information**

Land Zone 5						
3.5.5 (6 sites)	*Corymbia novoguinensis* +/- *C. tessellaris* woodland on sand plains on northern Cape York Peninsula	*Corymbia novoguinensis* (100%, 16)	*Corymbia novoguinensis* (40.1) *Corymbia tessellaris* (22.8) *Gardenia* sp. (19.5) *Welchiodendron longivalve* (12.2) *Acacia polystachya* (12.1) *Acacia crassicarpa (10.3)* *Bridelia tomentosa* (8) *Tabernaemontana orientalis* (6.9) *Mallotus nesophilus* (6.9) *Terminalia muelleri* (6.2) *Syzygium suborbiculare* (6.1) *Livistona muelleri* (6)	*Corymbia novoguinensis* (40.1) *Corymbia tessellaris* (22.8) *Gardenia* sp. (19.5) *Dodonaea polyandra* (13.1) *Welchiodendron longivalve* (12.2) *Acacia polystachya* (12.1) *Acacia crassicarpa (10.3)* *Bridelia tomentosa* (8) *Tabernaemontana orientalis* (6.9) *Mallotus nesophilus* (6.9) *Terminalia muelleri* (6.2) *Syzygium suborbiculare* (6.1) *Livistona muelleri* (6)	*Welchiodendron longivalve* (17%, 12) *Corymbia tessellaris* (50%, 11) *Corymbia nesophila* (13%, 10)	Occurs in the northern Peninsula area and Torres Strait Islands.
3.5.6 (4 sites)	*Eucalyptus phoenicea* woodland on sandy outwash plains	*Eucalyptus phoenicea* (100%, 17)	*Eucalyptus phoenicea* (41.8) *Eucalyptus portuensis* (15.4)	*Eucalyptus phoenicea* (41.8) *Eucalyptus portuensis* (15.4) *Xanthorrhoea*	*Eucalyptus portuensis* (25%, 10) *Grevillea glauca* (100%, 2) T2	On the Battlecamp sandstone plateau. *Eucalyptus portuensis* can be co-dominant

			Melaleuca nervosa (7.3)	*johnsonii* (14.7) *Melaleuca nervosa* (7.3) *Acacia torulosa* (6.8)	*Xanthorrhoea johnsonii* (25%, 15) S1	
3.5.9 (11 sites)	*Eucalyptus tetrodonta*, *Corymbia stockeri* +/- *C. setosa* woodland on sand plains	*Erythrophleum chlorostachys (*100%, 6*)* *Eucalyptus tetrodonta* (100%, 4) *Corymbia stockeri* (93%, 3)	*Corymbia setosa* (17.8) *Neofabricia sericisepala* (12.8) *Erythrophleum chlorostachys* (10.2) *Melaleuca nervosa* (6.1)	*Corymbia setosa* (17.8) *Neofabricia sericisepala* (12.8) *Erythrophleum chlorostachys* (10.2) *Melaleuca nervosa* (6.1)	*Corymbia setosa* (64%, 4) *Corymbia novoguinensis* (9%, 9)	Widespread in the southern half of the bioregion and occurs predominantly on the Holroyd Plain in close association with 3.5.37a. At the northern extent of this community *Corymbia novoguinensis* can dominate the canopy and *C. setosa* is absent. *Corymbia setosa* is not always present but there is often a combination of mixed species, including heaths, in the community. *C. stockeri* is likely to be subsp. *peninsularis.*

3.5.15b (1 site)	*Dapsilanthus spathaceus* open sedgeland with emergent shrubs.	*Dapsilanthus spathaceus* (100%, 19) *Germainia capitata* (100%, 10) *Schoenus sparteus* (100%, 9) *Scleria* sp. (100%, 6) *Pandanus* sp. (100%, 6)	*Dapsilanthus spathaceus* (42.7) *Germainia capitata* (30.9) *Schoenus sparteus* (29.3) *Scleria* sp. (23.9) *Tricostularia undulata* (13.8) *Eriocaulaceae* (9.7) *Xyris* sp. (9.7)	*Dapsilanthus spathaceus (42.7)* *Germainia capitata (30.9)* *Schoenus sparteus (29.3)* *Scleria sp. (23.9)* *Pandanus (22.1)* *Poaceae (19.5)* *Tricostularia undulata (13.8)* *Eriocaulaceae (9.5)* *Xyris sp. (9.5)*		Occurs in mosaic patches with 3.5.15a on Torres Strait Islands
3.5.19 (14 sites)	*Asteromyrtus lysicephala* and/or *Neofabricia myrtifolia* and/or *Jacksonia thesioides* open heath to shrubland on sand sheets	*Asteromyrtus lysicephala* (86%, 8) *Jacksonia thesioides* (86%, 3) *Neofabricia myrtifolia* (86%, 5) *Choriceras tricorne* (71%, 3) *Hibbertia banksii* (71%, 1) *Allocasuarina littoralis* (79%, 3) E.	*Asteromyrtus lysicephala* (24.3) *Allocasuarina littoralis* (12.8) *Thryptomene oligandra* (11.5) *Neofabricia myrtifolia* (11.3) *Hibbertia banksii* (8.9) *Leucopogon ruscifolius* (8.8) *Choriceras tricorne* (8.4) *Gompholobium nitidum* (8.2) *Grevillea*	*Asteromyrtus lysicephala* (24.3) *Allocasuarina littoralis* (12.8) *Thryptomene oligandra* (11.5) *Neofabricia myrtifolia* (11.3) *Hibbertia banksii* (8.9) *Leucopogon ruscifolius* (8.8) *Choriceras tricorne* (8.4) *Gompholobium nitidum* (8.2) *Grevillea pteridifolia* (6.8)	*Leucopogon ruscifolius* (29%, 4) *Thryptomene oligandra* (21%, 7)	Heath formation differentiates this from RE 3.5.42 which is a low woodland to open forest.

			pteridifolia (6.8)			
3.5.25 (1 site)	*Eucalyptus leptophleba* woodland on plains	*Eucalyptus leptophleba* (100%, 21) *Eucalyptus chlorophylla* (100%, 2) *Corymbia clarksoniana* (100%, 2)	*Eucalyptus chlorophylla* (6.3)	*Eucalyptus leptophleba* (42.4) *Eucalyptus chlorophylla* (6.3)		
3.5.34 (4 sites)	*Corymbia nesophila* open forest on sand rises in the Torres Strait	*Corymbia nesophila* (100%, 20)	*Corymbia nesophila* (22.5) *Acacia simsii* (9.7)	*Corymbia nesophila* (22.5) *Acacia simsii* (9.7)	*Corymbia clarksoniana* (50%, 13) *Corymbia stockeri* (50%, 6) *Jacksonia thesioides* (50%, 15) S1	Occurs on Torres Strait Islands. *Corymbia stockeri* is likely to be *Corymbia stockeri* subsp. *peninsularis.*

3.5.35 (13 sites)	*Eucalyptus tetrodonta, Corymbia nesophila* woodland with a heathy understory	*Eucalyptus tetrodonta* (100%, 8) *Corymbia nesophila* (92%, 5) *Asteromyrtus brassii* (77%, 4) T2. *Neofabricia myrtifolia* (69%, 4) T2 *Neoroepera banksii* (69%, 6) S1 *Acacia calyculata* (77%, 2) S1	*Neoroepera banksii* (14.1) S1 *Asteromyrtus brassii* (8) T2 *Acacia calyculata (6.7) S1* *Neofabricia myrtifolia (6.3)*	*Neoroepera banksii* (14.1) S1 *Lophostemon suaveolens* (10.3) *Asteromyrtus brassii* (8) T2 *Acacia calyculata* (6.7) S1 *Neofabricia myrtifolia* (6.3)	*Lophostemon suaveolens* (15%, 13)	Characterised by a dominance of heath species in the understory. The understory species mix of this community may change depending on fire history.
3.5.36	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland to open forest on undulating plains and remnant plateaus					
3.5.36a (40 sites)	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland on undulating plains	*Eucalyptus tetrodonta* (100%, 11) *Corymbia nesophila* (100%, 10) *Erythrophleum chlorostachys* (45%, 2)	*Corymbia nesophila* (13)*Eucalyptus tetrodonta* (7.6)	*Corymbia nesophila* (13)*Eucalyptus tetrodonta* (7.6)		Occurs on sandy and loamy plains off remnant bauxite plateaus.

3.5.36b (10sites)	*Eucalyptus tetrodonta* and *Corymbia nesophila* open forest on remnant plateaus	*Eucalyptus tetrodonta* (100%, 15) *Corymbia nesophila* (100%, 15) *Erythrophleum chlorostachys* (50%, 3)	*Corymbia nesophila* (19.7) *Planchonia careya* (6.5) T3, S1	*Corymbia nesophila* (19.7) *Eucalyptus tetrodonta* (12.3) *Parinari nonda* (6.8) *Planchonia careya* (6.5) T3, S1		Occurs on bauxite plateaus. May occur as a woodland.
3.5.37	*Eucalyptus tetrodonta +/- Corymbia stockeri* woodland to tall open forest on erosional plains and remnant plateaus					
3.5.37a (14 sites)	*Eucalyptus tetrodonta +/- Corymbia stockeri* woodland on erosional plains	*Eucalyptus tetrodonta* (100%, 12) *Corymbia stockeri* (93%, 7) *Erythrophleum chlorostachys* (43%, 2)	*Corymbia stockeri* (9.7) *Eucalyptus tetrodonta* (6.9)	*Corymbia stockeri* (9.7 *Eucalyptus brassiana* (7) *Eucalyptus tetrodonta* (6.9)	*Xanthorrhoea johnsonii* (21%, 5) S1	Occurs predominantly on erosional plains and rises but can occur on edges of bauxite plateaus. *Corymbia stockeri* is likely to be subsp. *peninsularis*.
3.5.37b (16 sites)	*Eucalyptus tetrodonta* +/- *Corymbia stockeri* tall woodland on tertiary remnant plateaus	*Eucalyptus tetrodonta* (100%, 18) *Corymbia stockeri* (81%, 9) *Erythrophleum*	*Corymbia stockeri* (12.6) *Erythrophleum chlorostachys* (11.6)	*Eucalyptus tetrodonta* (14.2) *Corymbia stockeri* (12.6) *Erythrophleum chlorostachys* (11.6)		Occurs predominantly on the Kimba plateau and northern extensions. *Corymbia stockeri* is likely to be subsp. *peninsularis*.

		chlorostachys (69%, 7)				
3.5.38	*Eucalyptus tetrodonta +/- E. cullenii* or *E. tetrodonta +/- C. stockeri and Melaleuca* spp. woodland on remnant surfaces					
3.5.38a (7 sites)	*Eucalyptus tetrodonta, Corymbia stockeri +/- E. cullenii* woodland on remnant surfaces	*Eucalyptus tetrodonta* (100%, 9) *Corymbia stockeri* (100%, 4) *Eucalyptus cullenii* (86%, 6)	*Eucalyptus cullenii* (25)	*Eucalyptus cullenii* (25) *Corymbia disjuncta* (7.5)		Both subspecies of *Corymbia stockeri* may occur. Occurs on weathered remnant surfaces often with ironstone nodules to surface. *Melaleuca stenostachya* may occur but will have lower biomass than the *C. stockeri* and *Eucalyptus cullenii.*
3.5.38b (7 sites)	*E. tetrodonta +/- C. stockeri* woodland with a *Melaleuca* spp shrub layer on remnant surfaces	*Eucalyptus tetrodonta* (100%, 8) *Corymbia stockeri* (86%, 7) *Melaleuca viridiflora* (100%, 3) T2 *Melaleuca*	*Corymbia stockeri* (7.9) *Melaleuca stenostachya* (7.9)	*Corymbia stockeri* (7.9) *Melaleuca stenostachya* (7.9)	*Jacksonia thesioides* (14%, 5) S1	Occurs on weathered remnant surfaces often with ironstone nodules to surface.

		stenostachya (43%, 4) *Corymbia clarksoniana* (43%, 2) *Melaleuca stenostachya* (57%, 4) T2			
3.5.39 (26 sites)	*Eucalyptus tetrodonta* +/- *Corymbia clarksoniana* woodland on sand plains	*Eucalyptus tetrodonta* (100%, 13) *Corymbia clarksoniana* (73%, 4) *Erythrophleum chlorostachys* (54%, *4)*	*Eucalyptus tetrodonta* (7.1)	*Eucalyptus tetrodonta* (7.1)	*Corymbia stockeri* (15%, 7) *Corymbia tessellaris* (12%, 7) *Eucalyptus leptophleba* (4%, 8)
3.5.40 (5 sites)	*Melaleuca stenostachya* +/- *Eucalyptus chlorophylla woodland* +/- *M. viridiflora* shrub layer on outwash plains	*Melaleuca stenostachya* (100%, 9) *Eucalyptus chlorophylla* (60%, 8) *Melaleuca viridiflora* (60%, 3)	*Melaleuca stenostachya* (35.7) *Eucalyptus chlorophylla (*17.3) *Corymbia hylandii* (8.7)	*Melaleuca stenostachya* (35.7) *Eucalyptus chlorophylla* (17.3) *Corymbia hylandii* (8.7) *Acacia brassii* (7.2)	
3.5.41	*Melaleuca viridiflora* +/- *Corymbia clarksoniana* woodland to low open woodland on plains				

3.5.41a (14 sites)	*Melaleuca viridiflora* +/- *Corymbia clarksoniana* woodland on plains	*Corymbia clarksoniana* (86%, 12) *Melaleuca viridiflora* (50%, 9) *Melaleuca viridiflora* (79%, 2) T2	*Corymbia clarksoniana* (17.7) *Melaleuca viridiflora (8.7)*	*Corymbia clarksoniana* (17.7) *Melaleuca viridiflora (8.7)*	*Melaleuca viridiflora* (64%, 2) S1	
3.5.41b (1 site)	*Melaleuca viridiflora* +/- *Petalostigma banksii* low open woodland on plains	*Melaleuca viridiflora* (100%, 7) *Petalostigma banksii* (100%, 5) T2	None > 6	*Petalostigma banksii* (14.6) *Melaleuca viridiflora* (10.3) *Wrightia saligna* (6.1)		
3.5.42 (14 sites)	*Asteromyrtus brassii* and/or *Neofabricia myrtifolia* low open forest to woodland on sand plains	*Neofabricia myrtifolia* (93%, 9) *Asteromyrtus brassii* (93%, 6) *Allocasuarina littoralis* (71%, 3) T2	*Leucopogon yorkensis* (24.1) *Neofabricia myrtifolia* (20.3) *Asteromyrtus brassii* (19.4) *Callitris intratropica* (16.9) *Choriceras tricorne* (11.4) *Allocasuarina littoralis* (11.2) *Dodonaea polyandra* (10.6) *Welchiodendron longivalve* (10)	*Leucopogon yorkensis* (24.1) *Neofabricia myrtifolia* (20.3) *Asteromyrtus brassii* (19.4) *Callitris intratropica* (16.9) *Choriceras tricorne* (11.4) *Allocasuarina littoralis* (11.2) *Dodonaea polyandra* (10.6) *Welchiodendron longivalve* (10)	*Welchiodendron longivalve* (36%, 5) *Leucopogon yorkensis* (29%, 21) *Choriceras tricorne* (43%, 4) *Dodonaea polyandra* (36%, 4) *Callitris intratropica* (21%, 5) *Alyxia spicata* (71%, 2) S1 *Neoroepera banksii* (21%, 5) S1 *Melaleuca viridiflora* (50%, 5)	Woodland to open forest structure differentiates this from RE 3.5.19.

			Alyxia spicata (9.2) *Olax pendula* (6.8)	*Alyxia spicata* (9.2) *Olax pendula* (6.8)	
Land Zone 12					
3.12.7 (2 sites)	*Corymbia clarksoniana, Eucalyptus brassiana* open forest on granite ranges	*Corymbia clarksoniana* (100%, 20) *Eucalyptus brassiana* (100%, 12)	*Chionanthus ramiflorus* (29.4) *Eucalyptus brassiana* (29.1) *Calophyllum sil* (15.5) *Wikstroemia indica* (13.8) *Celtis paniculata* (12) *Polyscias elegans* (12) *Ganophyllum falcatum* (11.9) *Litsea glutinosa* (11.9) *Cupaniopsis anacardioides (6.9)*	*Corymbia clarksoniana* (35.5) *Chionanthus ramiflorus* (29.4) *Eucalyptus brassiana* (28.9) *Melaleuca nervosa* (21.8) *Calophyllum sil* (15.5) *Wikstroemia indica* (13.8) *Celtis paniculata* (12) *Polyscias elegans* (12) *Ganophyllum falcatum* (11.9) *Litsea glutinosa* (11.9) *Schefflera actinophylla* (11.9) *Buchanania arborescens* (10.9) *Wodyetia bifurcata* (9.8) *Cupaniopsis anacardioides* (6.9) *Alyxia spicata* (6.6)	*Corymbia stockeri* (50%,5) *Chionanthus ramiflorus* (50%, 20) T2 *Melaleuca nervosa* (50%, 12) T2

3.12.9 (1 site)	*Corymbia tessellaris* +/- *Welchiodendron longivalve* +/- *Eucalyptus cullenii* open forest on footslopes of granite hills.	*Corymbia tessellaris* (100%, 30)	*Tabernaemontana pandacaqui* (6.9)	*Corymbia tessellaris* (51.1) *Tabernaemontana pandacaqui* (6.9)		
3.12.10	*Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland or *E. chlorophylla* woodland on granitic ranges					
3.12.10a (13 sites)	*Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland on granitic ranges	*Eucalyptus cullenii* (100%, 12) *Corymbia clarksoniana* (69%, 5)	*Eucalyptus cullenii* (28.8) *Dendrolobium umbellatum* (14.8)	*Eucalyptus cullenii* (28.8) *Dendrolobium umbellatum* (14.8) *Dolichandrone heterophylla* (8.1) *Acacia oraria* (6.1)	*Dendrolobium umbellatum* (38%, 8) T2 *Dendrolobium umbellatum* (38%, 7) S1	This vegetation community is unmappable at the RE mapping scale and interspersed with vegetation community 3.12.10b. These communities are mapped collectively as 3.12.10
3.12.10b (2 sites)	*Eucalyptus chlorophylla* woodlands on granitic ranges	*Eucalyptus chlorophylla* (100%, 4) *Corymbia dallachiana* (100%, 1)	*Eucalyptus chlorophylla* (19.6) *Corymbia confertiflora* (9.6)	*Eucalyptus chlorophylla* (19.6) *Corymbia confertiflora* (9.5)	*Corymbia confertiflora* (50%, 3)	This vegetation community is unmappable at the RE mapping scale and interspersed with vegetation community 3.12.10a. These communities

						are mapped collectively as 3.12.10.
3.12.11 (6 sites)	*Corymbia stockeri* +/- *Welchiodendron longivalve* woodland on steep to rolling granite hills	*Corymbia stockeri* (100%, 23)	*Corymbia stockeri* (46.2) *Dalbergia densa* (10.9) *Welchiodendron longivalve* (9.4)	*Corymbia stockeri* (45.9) *Dalbergia densa* (10.9) *Cryptocarya exfoliata* (9.8) *Welchiodendron longivalve* (9.2)	*Eucalyptus cullenii* (17%, 10) *Corymbia tessellaris* (17%, 8) *Welchiodendron longivalve* (17%, 8) *Dalbergia densa* (83%,2) S1	
3.12.18	*Eucalyptus leptophleba* and *Corymbia clarksoniana* woodland or *Corymbia disjuncta* woodland on steep to low igneous hills					
3.12.18a (7 sites)	*Eucalyptus leptophleba* +/- *Corymbia clarksoniana* woodland to open woodland on steep to low igneous hills	*Eucalyptus leptophleba* (100%, 13) *Corymbia clarksoniana* (86%, 3)	*Eucalyptus leptophleba* (35.2) *Dendrolobium umbellatum* (10) *Flueggea virosa* (6.7)	*Eucalyptus leptophleba* (35.1) *Dendrolobium umbellatum* (9.9) *Eucalyptus platyphylla* (8.4) *Flueggea virosa* (6.7)	*Eucalyptus platyphylla* (14%, 8) *Dendrolobium umbellatum* (43%, 7) T2 *Dendrolobium umbellatum* (43%, 3) S1	
3.12.18b (2 sites)	*Corymbia disjuncta* +/- *C. clarksoniana*	*Corymbia disjuncta* (100%, 8) *Corymbia*	*Corymbia disjuncta* (29.4)	*Corymbia disjuncta* (29.4) *Piliostigma*		Restricted to Orchid Creek area.

	woodland on steep igneous hillslopes	*clarksoniana* (100%, 1)	*Piliostigma malabaricum* (6.9)	*malabaricum* (6.9) *Alstonia* sp. (6.3)		
3.12.28 (5 sites)	*Leptospermum purpurascens* tall shrubland on igneous hills	*Leptospermum purpurascens* (100%, 19)	*Leptospermum purpurascens* (44.3) *Shonia tristigma* (10.7) *Hovea* sp. (7.7)	*Leptospermum purpurascens* (44.4) *Shonia tristigma* (10.7) *Acacia brassii* (10.2) *Hovea* sp.(7.7) *Lomandra banksii* (6.2)	*Shonia tristigma* (20%, 6) S2	
3.12.30 (1 site)	*Imperata cylindrica +/- Heteropogon contortus +/- Mnesithea rottboellioides* closed tussock grassland on steep slopes of igneous hills	*Imperata cylindrica* (100%, 52) *Heteropogon contortus* (100%, 18) *Mnesithea rottboellioides* (100%, 11)	*Imperata cylindrica* (71.3) *Heteropogon contortus* (41.6) *Mnesithea rottboellioides* (32.5) *Cymbopogon refractus* (19.6)	*Imperata cylindrica* (71.4) *Heteropogon contortus* (41.6) *Mnesithea rottboellioides* (32.5) *Cymbopogon refractus* (19.6)		
3.12.32(2 sites)	*Schizachyrium* spp. +/- *Rhynchosia* spp. grasslands on granite hills and rock pavements	*Schizachyrium* sp. (100%, 9)*Ectrosia* sp. (100%, 8)*Rhynchosia minima* (100%, 5)*Eriachne* sp. (100%, 4)	*Ectrosia leporina* (26.8) *Rhynchosia* sp. (19.8) *Eriachne* sp. (18.1) *Schizachyrium* sp. (13.8) *Polycarpaea spirostylis* (12.2)	*Ectrosia leporina* (26.9) *Rhynchosia sp.* (19.7) *Eriachne sp.* (18.1) *Schizachyrium sp.* (13.6) *Polycarpaea spirostylis* (12.2) *Scleria* sp. (10.5) *Fimbristylis* sp.(9.4)		Restricted to granite hills south of Coen.

			Scleria sp. (10.5) *Fimbristylis* sp. (9.4)	*Pseudopogonatherum contortum* (6.9) *Ptilotus* sp. (6.9) *Indigofera* sp. (6.7)	
3.12.34 (2 sites)	Igneous rock pavements associated with mountains and some offshore islands	*Rhynchosia* sp. (100%)	Rock pavement (99.5)	Rock pavement (99.5)	Occurs as a complex of rock pavements (rather than boulder piles) with shrubland that can include *Acacia umbellata*, *Canarium australianum*, *Psydrax* spp., *Terminalia* spp. and *Carissa ovata* growing in areas where soil accumulates
3.12.39	*Eucalyptus crebra* +/- *Corymbia hylandii* or *Lophostemon suaveolens* low woodland to low open forest on granite hills				

3.12.39a (2 sites)	*Eucalyptus crebra* +/- *Corymbia hylandii* low woodland to low open forest on skeletal soils on gullies and foothills of granite hills	*Eucalyptus crebra* (100%, 10)	*Eucalyptus crebra* (22.2) *Diospyros* sp.(19.1) *Acacia disparrima* (17.7) *Manilkara kauki* (16.9) *Terminalia subacroptera* (14.7) *Micromelum minutum* (10) *Alphitonia excelsa* (9.8) *Argophyllum verae* (9.8) *Bombax ceiba* (9.8) *Cupaniopsis fleckeri* (9.8) *Psychotria* sp. *(7.2)*	*Corymbia hylandii* (29.8) *Cochlospermum gillivraei* (27.4) *Eucalyptus crebra* (21.9) *Diospyros* sp. (19.1) *Acacia disparrima* (17.7) *Manilkara kauki* (16.9) *Terminalia subacroptera* (14.7) *Gardenia* sp. (12.8) *Erythrophleum chlorostachys* (11.2) *Micromelum minutum* (10) *Alphitonia excelsa* (9.8) *Argophyllum verae* (9.8) *Bombax ceiba* (9.8) *Cupaniopsis fleckeri* (9.8) *Psydrax* sp (8.8) *Sterculia quadrifida* (8) *Psychotria* sp. (7.2) *Euroschinus falcatus* (6.7)	*Corymbia hylandii* (50%, 19) *Cochlospermum gillivraei* (50%, 18)

3.12.39b (1 site)	*Lophostemon suaveolens* low open forest on upper slopes of granite ranges.	*Lophostemon suaveolens* (100%, 30) *Eucalyptus crebra* (100%, 8)	*Endiandra glauca* (53.2) *Myrsine variabilis* (21.9) *Litsea breviumbellata* (16.9) *Cyclophyllum coprosmoides* (13.8) *Rhodamnia* sp. (13.8) *Cryptocarya vulgaris (9.8)* *Guioa acutifolia (9.8)* *Mischocarpus* sp. *(9.8)*	*Endiandra glauca* (53.2) *Lophostemon suaveolens* (51.8) *Acacia flavescens* (30.3) *Myrsine variabilis* (21.9) *Litsea breviumbellata* (16.9) *Cyclophyllum coprosmoides* (13.8) *Muehlenbeckia zippelii* (13.9) *Rhodamnia sp. (13.9)* *Cryptocarya vulgaris* (9.8) *Guioa acutifolia* (9.8) *Mischocarpus* sp. *(9.8)*	*Endiandra glauca* (100%, 30) S1
3.12.40 (7 sites)	*Corymbia nesophila +/- Eucalyptus tetrodonta* woodlands on igneous hills and rises	*Corymbia nesophila* (100%, 27)	*Corymbia nesophila* (52) *Eucalyptus tetrodonta* (16.9) *Coelospermum reticulatum* (9.8)	*Corymbia nesophila* (51.7) *Eucalyptus tetrodonta* (16.6) *Neoroepera banksii* (10.6) *Coelospermum reticulatum* (9.8) *Antidesma ghaesembilla* (8.7) *Erythrophleum*	*Eucalyptus tetrodonta* (57%, 12)

				chlorostachys (7.5) *Hibbertia* sp. (7)	
3.12.41 (6 sites)	*Eucalyptus tetrodonta* woodland +/- heath species on granite hills and rises	*Eucalyptus tetrodonta* (100%, 6) *Neofabricia myrtifolia* (83%, 3) T2 *Asteromyrtus brassii* (67%, 3) T2 *Acacia rothii* (67%, 5) T2 *Xanthorrhoea johnsonii* (83%, 3) S1	*Asteromyrtus brassii* (19.1) *Acacia rothii* (17.8) *Neofabricia myrtifolia* (12.4) *Eucalyptus tetrodonta* (10.2) *Grevillea glauca* (10) *Xanthorrhoea johnsonii* (9.8) *Hibbertia banksii* (8.9) *Allocasuarina littoralis* (8.4) *Corymbia nesophila* (7.6) *Banksia dentata* (6.3)	*Asteromyrtus brassii* (19) *Acacia rothii* (17.7) *Neofabricia myrtifolia* (12.2) *Eucalyptus tetrodonta* (10) *Grevillea glauca* (9.9) *Xanthorrhoea johnsonii (9.7)* *Hibbertia banksii (8.9)* *Allocasuarina littoralis (8.3)* *Corymbia nesophila (7.4)* *Banksia dentata (6.3)*	*Corymbia nesophila* (50%, 8) *Asteromyrtus brassii* (50%, 7) *Corymbia stockeri* (17%, 8)
3.12.42(12 sites)	*Eucalyptus tetrodonta* woodland on low to undulating granite hills.	*Eucalyptus tetrodonta* (100%, 11) *Corymbia clarksoniana* (67%, 3)	*Eucalyptus tetrodonta* (20.7)	*Eucalyptus tetrodonta* (20.4) *Erythrophleum chlorostachys* (7.3)	*Eucalyptus brassiana* (8%, 17)*Corymbia stockeri (8%, 15)*
3.12.43 (5 sites)	*Welchiodendron longivalve, Acacia*				

	brassii low woodland on igneous hills					
3.12.43a (4 sites)	*Welchiodendron longivalve* and *Acacia brassi*i low woodland on undulating igneous rises to steep hills.	*Welchiodendron longivalve* (100%, 7) *Acacia brassii* (75%, 6)	*Welchiodendron longivalve* (24) *Acacia brassii* (17.4)	*Welchiodendron longivalve* (24) *Acacia brassii* (17.4) *Eucalyptus platyphylla* (13.5) *Bursaria incana* (8.3) *Petalostigma pubescens* (7.6) *Livistona muelleri (6.1)*	*Eucalyptus cullenii (25%, 13)* *Eucalyptus platyphylla* (25%, 10) *Neofabricia myrtifolia* (25%, 6)	
3.12.43b (1 site)	*Melaleuca viridiflora* and *Welchiodendron longivale* shrubland on rocky igneous headlands	*Corymbia tessellaris* (100%,10) *Jacksonia thesioides* (100%,6)	*Corymbia tessellaris* (20.4) *Jacksonia thesioides* (19.6)	*Corymbia tessellaris* (20.4) *Jacksonia thesioides* (19.6) *Acacia crassicarpa* (13.5) *Croton arnhemicus* (8.2) *Syzygium suborbiculare* (8.1)	*Corymbia tessellaris* (100%,6) E *Acacia crassicarpa* (100%,4) E	Restricted to continental Torres Strait islands.
3.12.44 (3 sites)	*Melaleuca citrolens* low open woodland on low granite hills and rolling rises	*Melaleuca citrolens* (100%, 3)	*Melaleuca citrolens* (24.7) *Melaleuca foliolosa* (11.3) *Petalostigma banksii* (8.9)	*Melaleuca citrolens* (24.7) *Melaleuca foliolosa* (11.3) *Petalostigma banksii* (8.8)	*Melaleuca foliolosa* (33% ,3) *Petalostigma banksii* (33%, 3)	

3.12.45 (5 sites)	*Melaleuca viridiflora* low woodland to low open woodland occurs on steep igneous hills and footslopes	*Melaleuca viridiflora* (100%, 13)	*Melaleuca viridiflora* (22.9)	*Melaleuca viridiflora* (22.9) *Choriceras tricorne* (8.1)	*Eucalyptus tetrodonta* (20%, 4) E *Choriceras tricorne* (20%, 13) S1
3.12.47	Mixed heath species low woodland to wetter dwarf shrubland on igneous hills				
3.12.47a (4 sites)	Mixed heath species tall shrubland on igneous hills	*Asteromyrtus lysicephala* (100%, 3) S1 *Melaleuca viridiflora* (40%, 6) *Allocasuarina littoralis* (40%, 4) *Neofabricia myrtifolia* (40%, 4) *Melaleuca arcana* (40%, 4) *Asteromyrtus brassii* (40%, 3)	*Allocasuarina littoralis* (17) *Choriceras tricorne* (11.1) *Asteromyrtus lysicephala* (10.6) *Xanthorrhoea johnsonii* (9.9) *Melaleuca arcana* (9.8) *Jacksonia thesioides (8.9)* *Neofabricia myrtifolia* (7.3)	*Allocasuarina littoralis* (17) *Choriceras tricorne* (11.1) *Asteromyrtus lysicephala* (10.6) *Xanthorrhoea johnsonii* (9.9) *Melaleuca arcana* (9.8) *Jacksonia thesioides (8.9)* *Neofabricia myrtifolia (7.3)*	
3.12.47b (1 site)	*Asteromyrtus lysicephala, Choriceras tricorne, Jacksonia thesioides* dwarf shrubland on igneous	*Asteromyrtus lysicephala* (100%, 6) *Choriceras tricorne* (100%, 5) *Jacksonia thesioides*	None	*Asteromyrtus lysicephala* (10.6) *Melaleuca viridiflora* (13.3)	

	slopes with impeded drainage	(100%, 4) *Melaleuca viridiflora* (100%, 3)		*Grevillea pteridifolia* (7)	
3.12.48	*Heteropogon triticeus* or *Themeda triandra* or *Schizachyrium fragile* tussock grassland on rocky igneous coastal headlands and islands				
3.12.48a (5 sites)	*Heteropogon triticeus +/- Sarga plumosum* grasslands on igneous headlands and offshore islands	*Heteropogon triticeus* (100%, 42) *Sarga plumosum* (80%, 14)	*Heteropogon triticeus* (63.4) *Sarga plumosum* (32.2) *Ipomoea* sp. (9.8) *Eragrostis* sp. (8.3)	*Heteropogon triticeus* (63.5) *Sarga plumosum* (32.2) *Lomandra* sp. (15.8) *Lepturus repens* (13.9) *Ipomoea* sp. (9.8) *Salsola australis* (9.8) *Sesuvium portulacastrum* (9.8) *Eulalia mackinlayi* (9.6) *Lithomyrtus obtusa* (9.5) *Bulbostylis barbata* (8.8) *Eragrostis* sp. (8.3)	*Lomandra* sp. (20%, 13) *Lepturus repens* (20%, 10) *Lithomyrtus obtusa* (20%,6) E

3.12.48c (1 site)	*Schizachyrium* spp., *Aristida* spp. grasslands on igneous headlands	*Aristida* sp. (100, 20) *Schizachyrium* sp. (100%, 20)	None	*Aristida* sp. (43.3) *Schizachyrium* sp. (35.7)

Appendix 4.2. Regional ecosystems and vegetation communities on land zone 5 and 12 in Cape York Peninsula bioregion

I have retained the RE numbering system in this appendix. Land zone 5 = Tertiary landscape, land zone 12 = igneous landscape. Short label descriptions only are included. For more detailed descriptions please visit the website https://environment.ehp.qld.gov.au/regional-ecosystems . Communities are grouped by vegetation formation.

RE Label	Short Description	Area km²) *(areas <1 km² in Ha)*	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
Land zone 5						
Grasslands (0.01% of land zone)						
3.5.15b	*Dapsilanthus spathaceu*s open sedgeland with emergent shrubs.	120 Ha		Quantitative	1	Occurs in mosaic patches with 3.5.15a only on Torres Strait Islands.
3.5.29	*Themeda triandra* and *Heteropogon contortus* closed tussock grasslands on erosional plains.	686 Ha	1	Qualitative	0	The grassland is a fire climax community, with scattered vine forest remnants clumped on low granite mounds which protrude above the undulating sand plain, offering some protection from fire. Moa Island in the Torres Strait and also on coastal areas and islands in the north-east of the bioregion.
Shrublands (4% of land zone)						
3.5.19	*Asteromyrtus lysicephala* and *Neofabricia myrtifolia* open heath to shrubland on sand sheets.	1,902	1	Quantitative	14	Heath formation differentiates this from RE 3.5.42 which is a low woodland to open forest. From Coen to

						the McHenry Uplands. Also occurs just north of Hopevale.
3.5.43	*Asteromyrtus brassii* + *Melaleuca saligna* tall shrubland on residual sand plains.	230 Ha	1	Qualitative	0	Restricted to Moa Island in the Torres Strait.
Woodlands (95% of land zone)						
3.5.5	*Corymbia novoguinensis* +/- *C. tessellaris* woodland on sand plains on northern Cape York Peninsula.	117	1	Quantitative	6	Occurs on northern Cape York Peninsula and Torres Strait islands.
3.5.6	*Eucalyptus phoenicea* +/- *E. tetrodonta* woodland on sandy outwash plains.	401	1	Quantitative	4	On the Battlecamp sandstone plateau.
3.5.9	*Eucalyptus tetrodonta* and *Corymbia stockeri* and/or *C. setosa* on sand plains.	5,346	1	Quantitative	11	Western side of Great Dividing Range and Kalpowar Plains. Widespread in the southern half of the bioregion and occurs predominantly on the Holroyd Plain in close association with 3.5.37a.
3.5.15	*Melaleuca viridiflora* and *Asteromyrtus symphyocarpa* low woodland on colluvial plains.	33	2			
Vegetation communities:						
3.5.15a	*Melaleuca viridiflora* +/- *M. saligna* +/- *Corymbia* spp. low woodland or tall shrubland on residual sands.	32		Qualitative	0	Occurs on depositional and erosional plains. Widespread throughout the bioregion.
3.5.15b (Included in grasslands)						

3.5.21	*Corymbia clarksoniana* +/- *C. tessellaris* open forest on coastal ranges and lowlands.	54	1	Qualitative	0	On east coast from Cooktown to Lockhart River.
3.5.24	*Eucalyptus chlorophylla* +/- *Corymbia clarksoniana* woodland on erosional plains.	290	1	Qualitative	0	Occurs in the south-east of the bioregion on the Laura Basin and south of Cooktown.
3.5.25	*Eucalyptus leptophleba* and *E. platyphylla* woodland on outwash plains.	45	1	Quantitative	1	South-east of bioregion.
3.5.26	*Eucalyptus platyphylla* +/- *Corymbia clarksoniana* woodland to open forest on flat wet plains.	22	1	Qualitative	0	Occurs in the south-east of the bioregion close to the Wet Tropics Bioregion boundary.
3.5.27	*Melaleuca citrolens* +/- *M. foliolosa* +/- *M. viridiflora* low open woodland on plains.	711 Ha	1	Qualitative	0	Restricted to Rinyirru (Lakefield) National Park (CYPAL) within the Laura Basin.
3.5.32	*Asteromyrtus brassii* +/- *Syzygium angophoroides* +/- *Acmena hemilampra* open forest on residual sand rises and sheets.	228 Ha	1	Qualitative	0	Restricted to Moa Island, Torres Strait.
3.5.34	*Corymbia nesophila* and *C. stockeri* open forest on sand rises in the Torres Strait	116 Ha	1	Quantitative	4	Restricted to Moa Island, Torres Strait.
3.5.35	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland with heathy understory on sand plains.	2,241	1	Quantitative	13	Areas dominated by *Corymbia nesophila* sometimes occur. The understory species mix of this community may change depending on fire history. Extensive on sandplains from the upper reaches of the Archer River

						and Temple Bay through the McHenry Uplands to Bamaga.
3.5.36	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland on undulating plains and remnant plateaus.	17,751	2			
	Vegetation communities:					
3.5.36a	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland on undulating plains.	9,880		Quantitative	40	Occurs from Cape Melville, across to the Kimba plateau and throughout the north of the bioregion.
3.5.36b	*Eucalyptus tetrodonta* and *Corymbia nesophila* woodland to open forest on plateaus.	7,870		Quantitative	10	Small unmappable areas of 3.5.36a occur sporadically throughout the range of this community. Occurs on the Weipa Plateau and other remnant bauxite plateaus. 3.5.36b is generally taller than 3.5.36a. Occurs more often as an open forest in the north.
3.5.37	*Eucalyptus tetrodonta* and *Corymbia stockeri* +/- *Erythrophleum chlorostachys* woodland on erosional plains and deep massive sands.	5,587	2			
	Vegetation communities:					
3.5.37a	*Eucalyptus tetrodonta* and *Corymbia stockeri* woodland.	4,585		Quantitative	15	Occurs on depositional and erosional plains. Widespread throughout the bioregion.
3.5.37b	*Eucalyptus tetrodonta* ± *Erythrophleum chlorostachys* ± *Corymbia stockeri* tall woodland.	998		Quantitative	13	Predominantly on the Kimba Plateau, but may occur on other tertiary remnant plateaus in small patches below the scale of mapping. Typical vegetation is on

						the Kimba Plateau in an area known as the Desert. Occurs on tertiary remnant plateaus whilst 3.5.37a does not. 3.5.37b is generally taller than 3.5.37a (above 22m).
3.5.38	*Eucalyptus tetrodonta* and *E. cullenii* woodland or *E. tetrodonta* and *C. stockeri* +/- *Melaleuca* spp. woodland on remnant surfaces.	275	2			
Vegetation communities:						
3.5.38a	*Eucalyptus tetrodonta* and *E. cullenii* +/- *Corymbia stockeri* woodland on remnant weathered surfaces.	269		Quantitative	6	Central plains on the Peninsula from Musgrave to the Torres Strait Islands.
3.5.38b	*Eucalyptus tetrodonta* and *Corymbia stockeri* often with *Melaleuca stenostachya*.	640 Ha		Quantitative	7	Occurs in the central Peninsula from Coen to Bramwell Junction.
3.5.39	*Eucalyptus tetrodonta* and *Corymbia clarksoniana* woodland on sand plains.	9,848	1	Quantitative	25	Mainly occurs on plains in the Laura basin, but also in the central west Peninsula and on some Torres Strait Islands.
3.5.40	*Melaleuca stenostachya* +/- *Eucalyptus chlorophylla* +/- *M. viridiflora* woodland on outwash plains.		1	Quantitative	5	Occurs throughout Cape York including the Torres Strait Islands.
3.5.41	*Melaleuca viridiflora* +/- *Corymbia clarksoniana* woodland on plains.	2,386	2			
Vegetation communities:						

3.5.41a	*Melaleuca viridiflora* +/- *Corymbia clarksoniana* woodland on plains	318 Ha		Quantitative	14	Central plains on the Peninsula from Musgrave to the Torres Strait Islands.
3.5.41b	*Melaleuca viridiflora* low open woodland +/- *Petalostigma banksii* on plains.	336 Ha		Quantitative	1	Occurs in the central Peninsula from Coen to Bramwell Junction.
3.5.42	*Asteromyrtus brassii* and/or *Neofabricia myrtifolia* low open forest to woodland on sand plains.	426	1	Quantitative	14	Occurs throughout Cape York including the Torres Strait Islands.
Land zone 12						
Grasslands (2% of land zone)						
3.12.30	*Imperata cylindrica* +/- *Mnesithea rottboellioides* closed tussock grassland on steep slopes of igneous hills.	37	1	Quantitative	1	Occurs mainly in the northern McIlwraith Range, but also near Temple Bay and on some Torres Strait Islands.
3.12.32	*Schizachyrium* spp. +/- *Rhynchosia* spp. grasslands on shallow soils on undulating granite hills.	10	1	Quantitative	2	Restricted to granite hills south of Coen.
3.12.34	Igneous rock pavements associated with mountains and some offshore islands.	66	1	Quantitative	2	McIlwraith Range, Iron Range and Altanmoui Range. Torres Strait Islands.
3.12.48	*Heteropogon triticeus* or *Themeda triandra* or *Schizachyrium fragile* tussock grassland on rocky igneous coastal headlands and islands.	20	3			

Vegetation communities:

3.12.48a	*Heteropogon triticeus* dominated grasslands on igneous headlands and offshore islands.	11		Quantitative	5	Occurs on Torres Strait Islands
3.12.48b	*Themeda triandra* tussock grassland on igneous headlands and islands.	533 Ha		Qualitative	0	Occurs on Torres Strait Islands and other offshore islands and headlands along the east coast.
3.12.48c	*Schizachyrium* spp. +/- *Aristida* spp. grasslands on igneous headlands.	356 Ha		Quantitative	1	Restricted to headlands near the northern most extent of Cape York Peninsula and some east coast islands.
Shrublands (12% of land zone)						
3.12.28	*Leptospermum purpurascens* tall shrubland on igneous hills.	58	1	Quantitative	5	Occurs on the western edge of Iron Range.
3.12.43b	*Melaleuca viridiflora* and *Welchiodendron longivalve* shrubland on rocky igneous headlands.	9a Ha		Quantitative	1	Occurs on most continental Torres Strait Islands.
3.12.46	*Melaleuca stenostachya* shrubland on exposed igneous headlands and hills.	52	1	Qualitative	0	Occurs on granite hills between Musgrave Roadhouse and Archer River crossing as well as on some Torres Strait Islands.
3.12.47b	*Asteromyrtus lysicephala, Choriceras tricorne, Jacksonia thesioides* dwarf shrubland on igneous slopes with impeded drainage			Quantitative	1	Occurs west of the Iron Range.
Woodlands (75% of land zone)						

3.12.7	*Eucalyptus brassiana* and *Corymbia clarksoniana* open forest on granite ranges.	103	1	Quantitative	2	Occurs on western McIlwraith Range and the Melville Range.
3.12.8	*Corymbia clarksoniana* +/- *C. tessellaris* open forest on coastal granite ranges and lowlands.	185	1	Qualitative	0	McIlwraith Range and Iron Range, with extensive patches on some Torres Strait islands.
3.12.9	*Corymbia tessellaris* +/- *Welchiodendron longivalve* +/- *E. cullenii* open forest on footslopes of granite hills.	49	1	Quantitative	1	East of McIlwraith Range and Torres Strait islands close to the Australian mainland.
3.12.10[#]	*Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland or *E. chlorophylla* woodland on granitic ranges.	1,677	2			
	Vegetation communities:					[#]This RE is mapped as 3.12.10 as the vegetation communities are unmappable at the RE mapping scale.
3.12.10a	*Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland on granite hills and footslopes.	25		Quantitative	13	Occurs along the length of the Great Dividing Range. Also in small patches on ranges from Cooktown to the Altamoui Range.
3.12.10b	*Eucalyptus chlorophylla* woodlands on granite hills.	273 Ha		Quantitative	2	Occurs along the length of the Great Dividing Range. Also in small patches on ranges from Cooktown to the Altamoui Range.
3.12.11	*Corymbia stockeri* +/- *Welchiodendron longivalve* woodland on steep to rolling granite hills.	137	1	Quantitative	6	Southern Torres Strait islands and near Lockhart River.

3.12.18	*Eucalyptus leptophleba* and *Corymbia clarksoniana* woodland or *C. disjuncta* woodland on steep to low igneous hills.	483	2			
	Vegetation communities:					
3.12.18a	*Eucalyptus leptophleba* and *Corymbia clarksoniana* woodland to open woodland on steep to low igneous hills.	481		Quantitative	7	Central Peninsula along the Great Dividing Range.
3.12.18b	*Corymbia disjuncta* woodland on steep igneous hillslopes.	153 Ha		Quantitative	2	Restricted to hills northeast of Archer River Roadhouse.
3.12.23	*Acacia brassii* low open forest on igneous hills.	18	1	Qualitative	0	Torres Strait islands.
3.12.37	*Eucalyptus platyphylla* +/- *Corymbia* spp. woodland to open forest on coastal igneous headlands and footslopes.	547 Ha	1	Qualitative	0	Restricted to Horn and Muralag Islands in the Torres Strait.
3.12.38	*Corymbia clarksoniana* and/or *C. nesophila* and/or *C. stockeri* low woodland on acid volcanic hills.	17	1	Qualitative	0	Torres Strait Islands.
3.12.39	*Eucalyptus crebra* +/- *Corymbia hylandii* or *Lophostemon suaveolens* low open forest to low woodland on skeletal soils on gullies and foothills of granite hills.	63	2			

	Vegetation communities:					
3.12.39a	*Eucalyptus crebra* +/- *Corymbia hylandii* low open forest on skeletal soils on gullies and foothills of granite hills.	52		Quantitative	2	Occurs on Cape Melville
3.12.39b	*Lophostemon suaveolens, Eucalyptus crebra* low open forest on upper slopes of granite ranges.	11		Quantitative	1	Occurs on the Altanmoui Range north of Cooktown.
3.12.40	*Corymbia nesophila* +/- *Eucalyptus tetrodonta* woodlands on igneous hills and rises.	368	1	Quantitative	7	Occurs from Musgrave to Temple Bay on the Coen-Yambo Inlier. Small areas west of Cooktown and on some Torres Strait Islands.
3.12.41	*Eucalyptus tetrodonta* +/- *Corymbia nesophila* woodland on igneous hills and rises.	120	1	Quantitative	6	Occurs mainly around Iron Range but also found south of Coen and north of Bamaga.
3.12.42	*Eucalyptus tetrodonta* +/- *Corymbia clarksoniana* woodland on low to undulating granite hills.	2,445	1	Quantitative	12	Occurs along the length of the Coen -Yambo Inlier from Musgrave to Portlands Rds. Also on ranges from north-west of Cooktown.
3.12.43	*Welchiodendron longivalve* and *Acacia brassii* low woodland on undulating igneous rises to steep hills.	14	2			
	Vegetation communities:					
3.12.43a	*Welchiodendron longivalve* and *Acacia brassii* low woodland on	13		Quantitative	4	Occurs near Wolverton and Temple Bay as well as on some Torres Strait Islands.

	undulating igneous rises to steep hills.					
	3.12.43b (Included in shrublands)					
3.12.44	*Melaleuca citrolens* low open woodland on low granite hills and rolling rises.	61	1	Quantitative	3	Occurs on granite ranges from Coen to Palmerville.
3.12.45	*Melaleuca viridiflora* +/- *Eucalyptus* spp. low woodland to low open woodland on steep hills and footslopes.	141	1	Quantitative	5	Occurs along the length of the Great Dividing Range from the Hann River up to Portlands Rds.
3.12.47[#]	Mixed heath species low woodland to wetter dwarf shrubland on igneous hills	352	2	Quantitative		
	Vegetation communities:					[#]This RE is mapped as 3.12.47 as the vegetation communities are unmappable at the RE mapping scale.
3.12.47a	Mixed heath species low woodland on igneous hills			Quantitative	4	
	3.12.47b (Included in shrublands)					

Photographic examples of representative communities identified in the new classification system for the Tertiary and igneous landscapes on CYP

These were originally included in the publication Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia, 18*, 29 - 71.

RE 3.5.19 *Asteromyrtus lysicephala* and *Neofabricia myrtifolia* open heath to shrubland on sand sheets.

RE3.5.36a *Eucalyptus tetrodonta* and *Corymbia nesophila* woodland on undulating plains.

3.5.41b *Melaleuca viridiflora* low open woodland +/- *Petalostigma banksii* on plains.

3.12.10a *Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland on granite hills and footslopes.

RE 3.12.28 *Leptospermum purpurascens* tall shrubland on igneous hills.

RE 3.12.48a *Heteropogon triticeus* dominated grasslands on igneous headlands and offshore islands.

Appendix 4.3: Additional analysis requested by the technical review committee and recommendations

The expert panel queried two communities identified by the numerical analysis, requesting further analysis. These were the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands and the *Eucalyptus tetrodonta, Corymbia stockeri* woodlands, both distributed across the extent of land zone 5. The final recommendations are discussed below.

Methods

I carried out the initial investigations with the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands, testing for differences in three attributes; canopy heights of the tallest layer, and floristic differences in the woody and ground layer vegetation (separately). I tested each attribute for differences between landform (Tertiary remnant plateaus and sand plains), soil colour (red, yellow, brown) and soil texture (sand and earth) as recorded on site pro-formas. I used the ANOSIM routine (Clarke and Gorley 2006) which has two outputs; an R statistic and a significance value. The R statistic generally lies between 0 (there is no difference between the groups) and 1 (there is no similarity between the groups) but negative values indicate the within group variation is larger than the between group variation. In the ground layer I firstly looked for distinct species assemblages using *n*MDS and visually assessed whether these were coincident with different landform, soil colours or soil texture using GIS overlay. To test for differences in canopy height I also used an unpaired t-test as well as the ANOSIM routine. Due to the results of these investigations in the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands, analysis requested by review panel for the *Eucalyptus tetrodonta, Corymbia stockeri* woodlands was limited to differences in canopy height between landform (again Tertiary remnant plateaus and sand plains) and soil colour (red earths versus all other colours).

Of the 50 sites in the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands, 32 contained data useful for ground layer analysis and 49 for soil analysis. There were 3 additional sites not included in the original dataset which contained enough information for testing canopy heights. This resulted in 53 sites in the canopy height analysis. There were 31 sites in the *Eucalyptus tetrodonta, Corymbia stockeri* woodlands.

Results

Eucalyptus tetrodonta, Corymbia nesophila woodlands

Floristic differences in woody vegetation layers.

There was no floristic difference between soil textures (*R = -0.05, p = 0.75*), soil colours (*R = 0.08, p = 0.14*) or landform (*R = 0.01, p = 0.44*). The negative R value for soil texture indicates that the floristic differences individually on the sandy soils and on the earth soils is greater than the floristic differences between these two soil types.

Floristic differences in ground layer vegetation.

The two-dimensional *n*MDS ordination showed two ground layer species assemblages, one dominated by *Heteropogon triticeus* and the other by *Schizachyrium* species (Figure 4.1), but with a lot of variability as evidenced by the high stress level (0.2). However, these assemblages were not significantly associated with either different soil textures *(R = 0.02, p = 0.40)*, different soil colours *(R = -0.08, p = 0.71)* or different landforms *(R = 0.04, p = 0.33)*. This was also supported by the GIS overlay where there was no clear alignment of these assemblages with different soils or landforms.

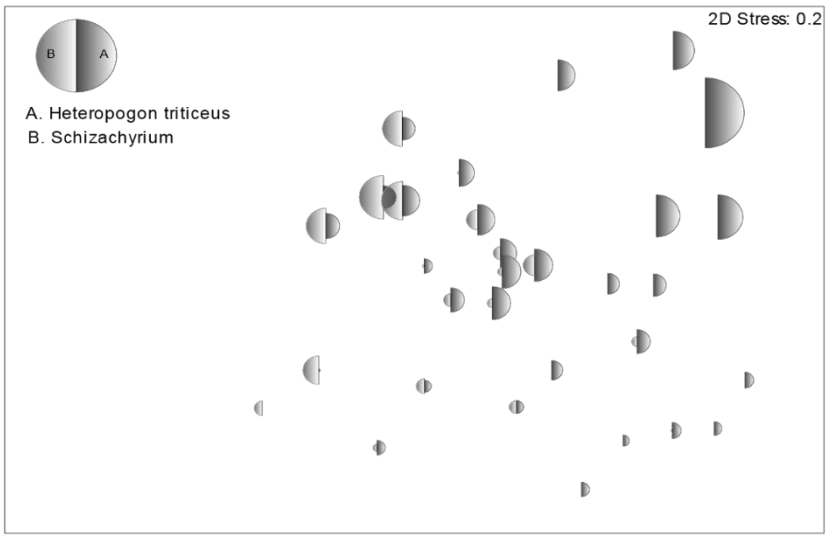


Figure 4.1 Bubble plot showing two species assemblages in the ground layer of the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands – one dominated by *Schizachyrium* spp, the other by *Heteropogon triticeus*. Abundances are standardised.

Canopy height differences

There was no difference in the canopy heights on different soil textures (*t(47) = 1.1, p = 0.28)* and the ANOSIM results indicated the variability of heights within individual soil textures was greater than between the soil textures (*R = -0.04*). Differences in canopy height on different coloured soils was not straight forward. There was a distinct, but not significant difference between the heights of trees on red earths versus brown earths (*R = 0.86, p = 0.06*), and an indistinct, but significant difference between the heights of trees on red earths versus yellow earths (*R = 0.18, p = 0.03*). The differences in canopy heights between landforms, however, was highly significant *(t(51) = 5.7, p<0.0001)*, with the average height of

trees on the Tertiary remnant plateaus being 5.2m taller than those on sand plains. I confirmed these results by running two different ANOSIM analysis. Firstly, I included all sites; 13 on the plateaus and 40 on the plains. These results showed a significant difference *(p = 0.01*), but a large overlap in height *(R = 0.25)*. I then ran ANOSIM with an equal number of sites (13) in both landforms (sites from the plains were chosen randomly). The difference in height was again significant *(p = 0.1)* however there was a small overlap in height *(R = 0.84)*.

Eucalyptus tetrodonta, Corymbia stockeri woodlands

There was a significant difference in the canopy heights of trees on both different landforms and different soil colours. The average height difference between trees on Tertiary remnant plateaus and on sand plains was 7.5m (*t(29) = 7.0, p<0.0001)* and on red earths versus all other coloured soil was 7.2m (*t(29) = 6.4, p<0.0001)*. The ANOSIM results show that there is overlap in tree height on both landform *(R = 0.63)* and soil colour *(R = 0.52)*.

Discussion

There are no differences in the floristics of the woody vegetation of the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands across land zone 5. There is a difference in the floristics of the ground layer, but it is not relatable to differences in soil types or landform and it is possible that the different assemblages are due to disturbance history (Kutt and Woinarski 2007, Miller and Murphy 2017).

There were significant differences in the canopy height of both *Eucalyptus tetrodonta, Corymbia nesophila* woodlands and *Eucalyptus tetrodonta, Corymbia stockeri* woodlands on different landforms and soil colour. The red earths, which are most common on the remnant plateaus, grow significantly taller woodlands than other coloured soils, which are most common on the sand plains. From this it is not surprising that the woodlands on the Tertiary remnant plateaus are significantly taller, however, as our ANOSIM results indicate there are areas on sand plains and on yellow earths where woodlands are also tall. This leads us to conclude that the height of woodlands on sand plains is variable, but woodlands on the remnant plateaus are consistently taller.

Recommendation

The classification criteria used in Queensland (Neldner et al. 2017) specify that woodlands with the same dominant species, but with a consistent height difference of >=5m, can be split into separate communities. Despite having no consistent floristic differences, the *Eucalyptus tetrodonta, Corymbia nesophila* woodlands and the *Eucalyptus tetrodonta, Corymbia stockeri* woodlands on the Tertiary remnant plateaus are consistently >=5m taller

than those on sand plains. However, there is an overlap in height between the plateaus and the sand plains. I therefore recommend the woodlands on the remnant plateaus are recognised as vegetation communities within the appropriate floristically defined regional ecosystem.

References

Clarke, K.R. and R.N. Gorley. 2006. *PRIMER v6: User Manual/Tutorial*. PRIMER-E, Plymouth.

Kutt, A.S. and J.C.Z. Woinarski. 2007. The effects of grazing and fire on vegetation and the vertebrate assemblage in a tropical savanna woodland in north-eastern Australia. *Journal of Tropical Ecology* 23 (1): 95-106.

Miller, B.P. and B.P. Murphy. 2017. Fire and Australian Vegetation. In: D. Keith (eds), *Australian Vegetation*. Cambridge University Press, Cambridge. 113-134.

Neldner, V.J., B.A. Wilson, H.A. Dilleward, T.S. Ryan and D.W. Butler. 2017. Methodology for Survey and Mapping of Regional Ecosystems and Vegetation Communities in Queensland. version 4. Queensland Herbarium, Queensland Department of Science, Information Technology and Innovation, Brisbane. https://publications.qld.gov.au/dataset/redd/resource/6dee78ab-c12c-4692-9842-b7257c2511e4 , accessed 1st June 2017.

Appendix 4.4: Floristic similarities between communities on land zone 5 and land zone 12 in Cape York Peninsula bioregion

I have retained the Regional Ecosystem numbering in this appendix. Plot data in each community was averaged. The dendrogram was formed using the CLUSTER routine and scatter plots using *n*MDS ordination in PRIMER v6 (Clarke & Gorley 2006).

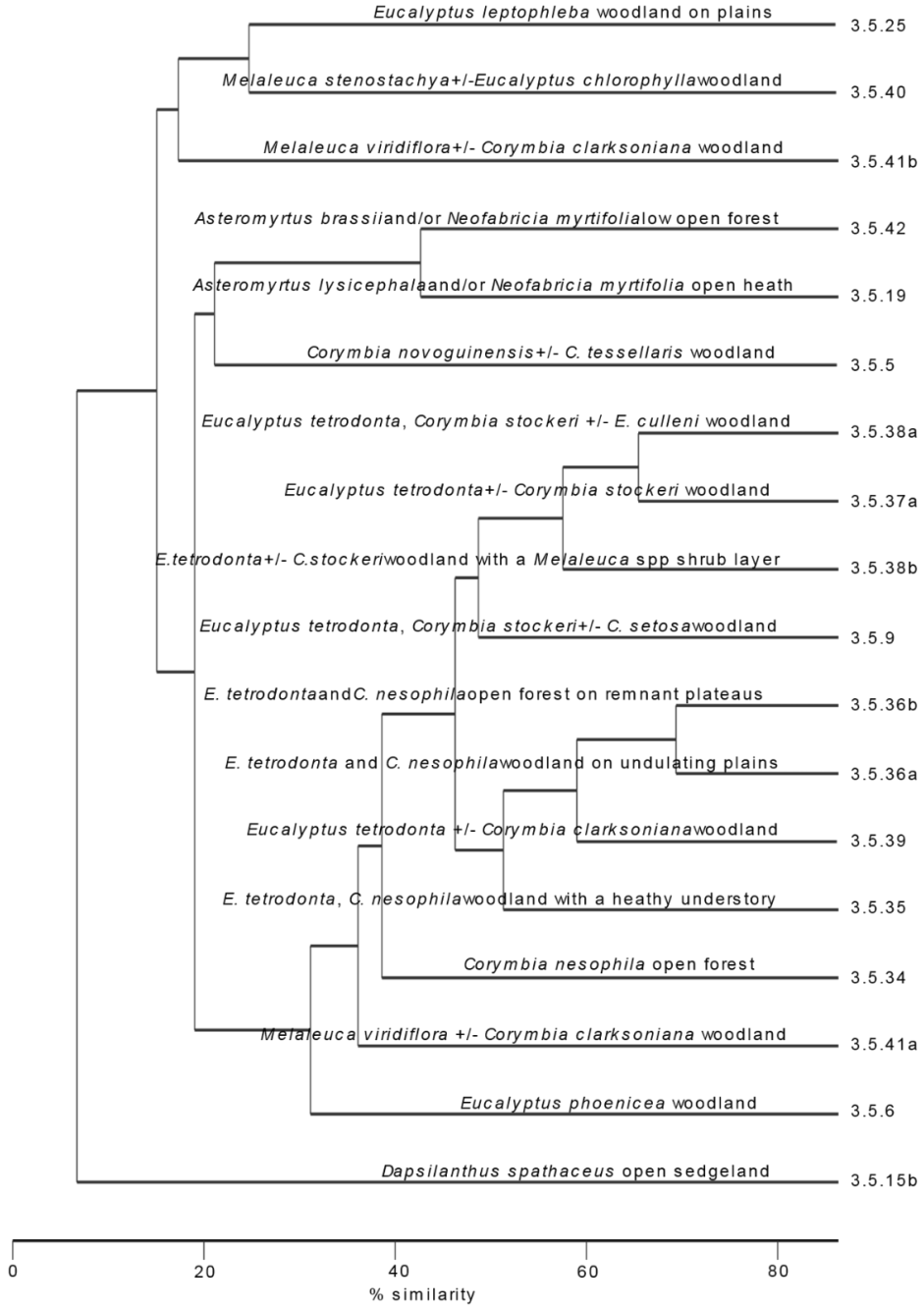


Figure A4.4 Dendrogram showing hierarchical relationships of communities on land zone 5 identified by quantitative analyses.

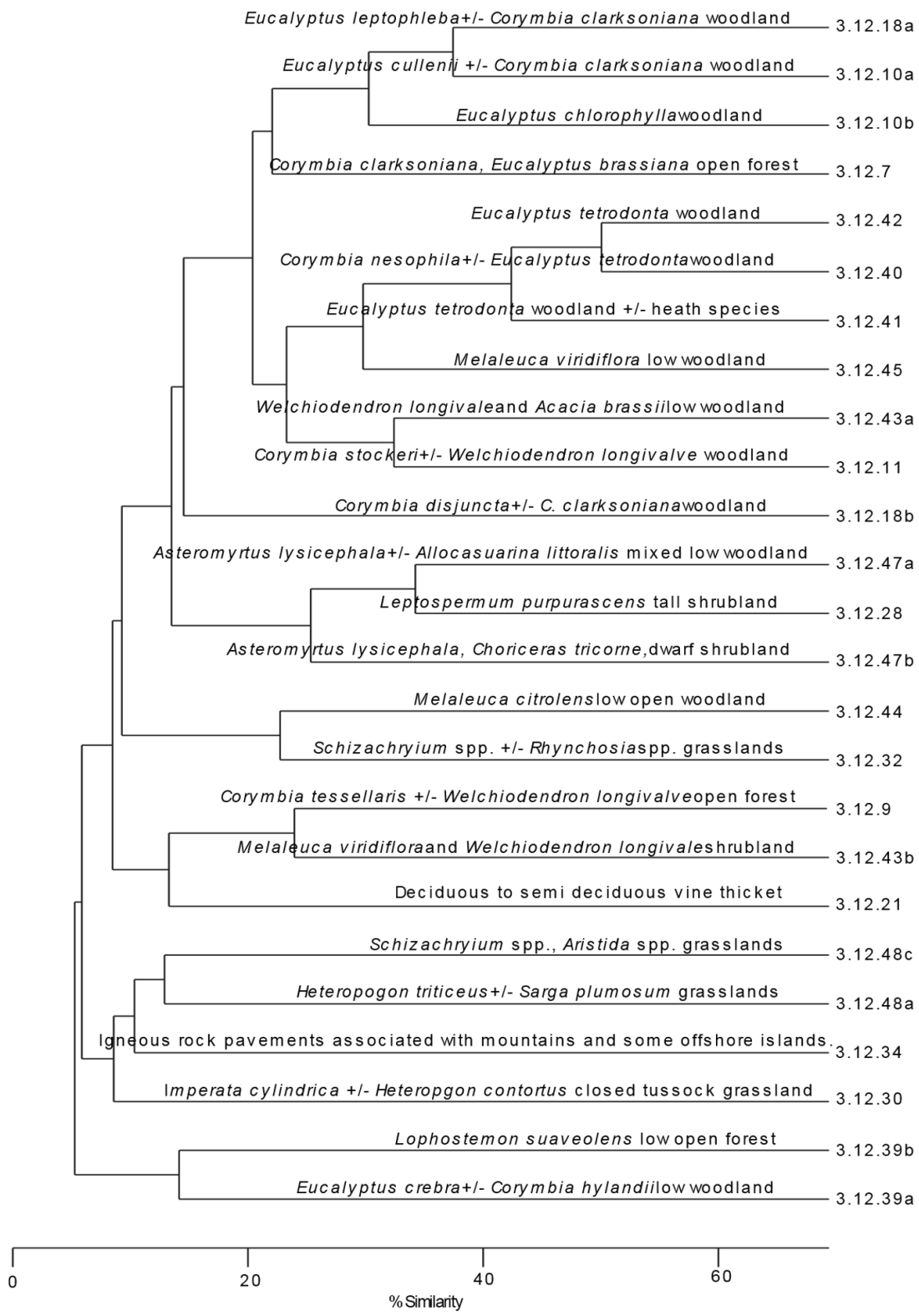


Figure A4.4: Dendrogram showing hierarchical relationships of communities identified by quantitative analyses on igneous landscape. Numbering of communities follows RE labelling format.

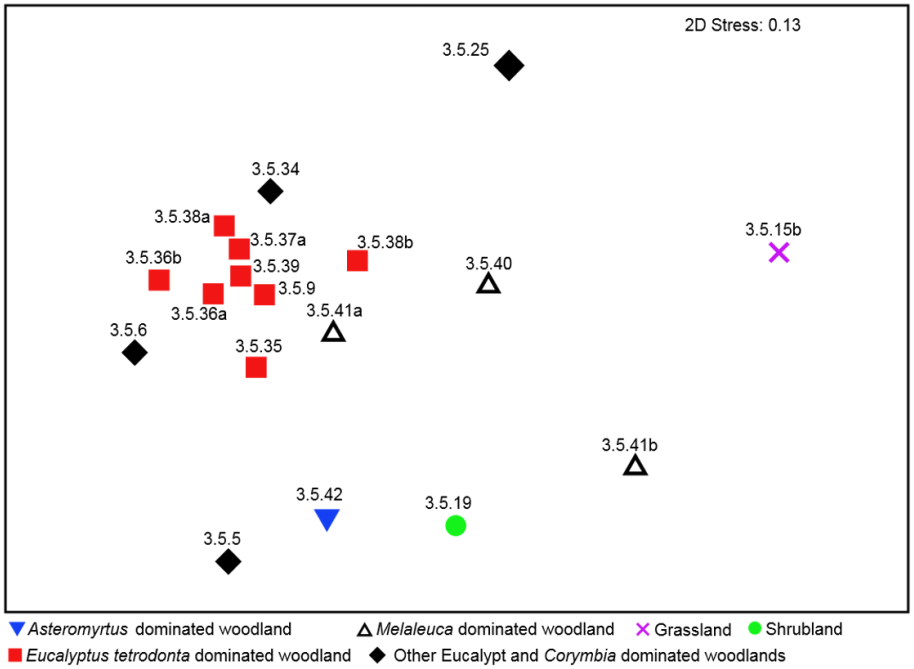


Figure A4.4: Land zone 5 scatter plot showing relative similarity of communities to each other in two-dimensions. Communities close together are more similar to each other. The greater clumping of communities than on land zone 12 scatter plot (Fig.5. 4) indicates a higher level of similarity of communities on land zone 5 than those on land zone 12.

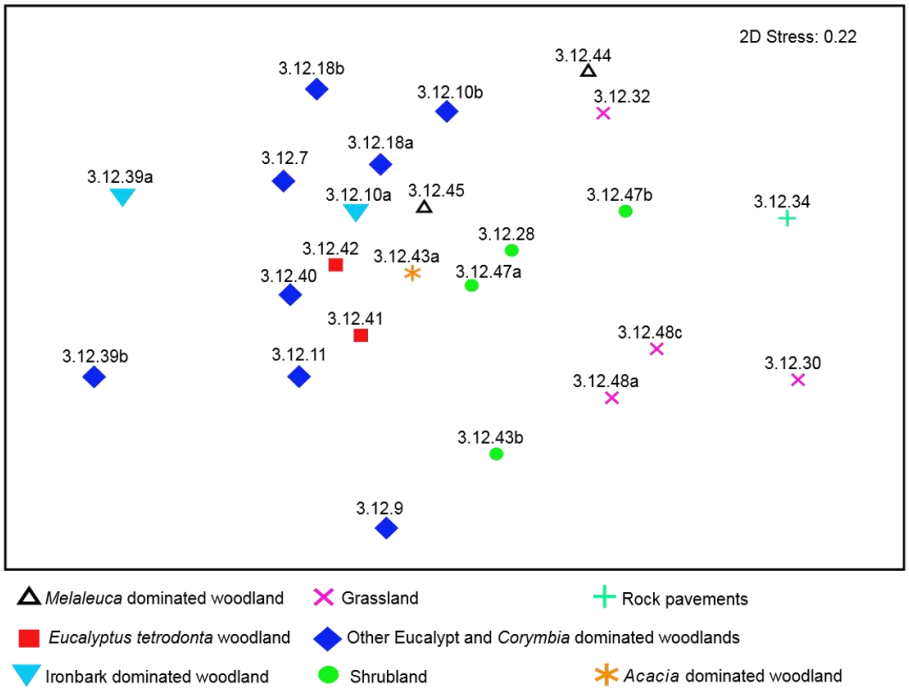


Figure A4.4: Land zone 12 scatter plot showing relative similarity of communities to each other. Communities close together are more similar to each other. The more scattered

spread of communities on land zone 12 when compared to the land zone 5 (Fig. 5.3) indicates a lower level of similarity between communities than land zone 5.

Table A4.4: Percent similarity matrix of communities on land zone 5 (calculated with Bray-Curtis coefficient).

	3.5.15b	3.5.19	3.5.25	3.5.34	3.5.35	3.5.36a	3.5.36b	3.5.37a	3.5.38a	3.5.38b	3.5.39	3.5.40	3.5.41a	3.5.41b	3.5.42	3.5.5	3.5.6	3.5.9
3.5.15b																		
3.5.19	9.2																	
3.5.25	8.4	7.6																
3.5.34	6.8	13.6	16.1															
3.5.35	6.6	36.5	9.7	38.8														
3.5.36a	5.9	14.8	9.6	47.8	56.5													
3.5.36b	3.0	9.5	7.5	44.6	50.9	69.4												
3.5.37a	6.0	12.4	15.0	36.2	45.2	61.7	47.8											
3.5.38a	4.4	10.1	17.8	38.7	42.7	52.8	41.3	65.4										
3.5.38b	8.5	16.2	17.3	38.1	37.5	38.7	30.3	53.7	61.4									
3.5.39	4.9	9.5	19.3	33.7	46.5	61.0	57.0	56.0	56.3	40.7								
3.5.40	8.0	20.8	24.7	20.4	18.6	18.1	13.2	22.1	25.1	49.6	17.6							
3.5.41a	7.6	16.6	24.1	32.2	34.7	33.5	28.9	35.0	35.9	35.8	46.9	26.8						
3.5.41b	14.0	19.1	10.6	5.9	6.9	7.3	3.6	7.7	6.5	23.7	10.5	24.1	25.6					
3.5.42	5.5	42.7	4.9	19.4	41.3	24.0	18.3	20.2	15.7	20.2	20.8	18.1	28.5	13.9				
3.5.5	4.5	11.2	5.1	22.0	26.6	25.4	23.9	12.6	9.6	13.3	24.5	10.4	26.6	13.3	31.0			
3.5.6	4.7	13.2	9.1	23.0	33.7	31.5	30.0	35.2	31.2	24.5	35.2	12.3	32.0	4.1	22.3	15.1		
3.5.9	5.9	10.1	13.9	30.9	42.0	46.8	44.2	53.1	49.2	43.8	56.1	20.3	42.1	14.2	17.8	17.1	35.0	

Table A4.4: Percent similarity matrix of communities on land zone 12 (calculated with Bray-Curtis coefficient).

	3.12.10a	3.12.10b	3.12.11	3.12.18a	3.12.18b	3.12.21	3.12.28	3.12.30	3.12.32	3.12.34	3.12.39a	3.12.39b	3.12.40	3.12.41	3.12.42	3.12.43b	3.12.43a	3.12.44	3.12.45	3.12.47a	3.12.47b	3.12.48a	3.12.48c	3.12.7
3.12.10a																								
3.12.10b	31.8																							
3.12.11	23.2	12.1																						
3.12.18a	37.4	28.8	9.5																					
3.12.18b	21.5	14.0	11.1	24.8																				
3.12.21	8.8	7.1	13.0	7.2	4.3																			
3.12.28	13.3	13.3	8.9	8.5	4.0	12.4																		
3.12.30	2.9	4.6	2.0	3.9	4.6	4.3	3.1																	
3.12.32	18.8	14.0	3.7	10.0	9.0	3.0	7.0	6.6																
3.12.34	2.3	4.9	2.7	3.2	5.2	4.7	4.4	7.0	7.9															
3.12.39a	14.8	2.8	6.3	4.7	12.5	4.0	9.5	2.0	1.9	2.4														
3.12.39b	5.1	2.1	6.4	2.7	5.1	2.9	2.8	2.4	2.2	3.1	14.1													
3.12.40	24.5	8.5	26.2	15.8	15.6	11.2	11.2	2.2	5.6	2.4	17.7	13.6												
3.12.41	24.6	7.0	23.3	15.6	3.5	12.9	16.2	2.2	8.5	2.1	2.6	10.0	44.2											
3.12.42	44.8	15.7	31.6	32.1	18.4	11.7	11.8	2.8	7.5	2.3	11.1	10.5	50.1	40.6										
3.12.43b	9.4	5.7	13.1	12.2	4.7	9.4	10.2	4.7	4.0	8.1	2.1	3.5	10.7	8.9	12.2									
3.12.43a	33.8	18.3	32.4	19.1	4.9	14.5	28.9	3.1	12.9	2.5	11.9	2.1	17.3	31.3	23.8	6.5								
3.12.44	13.1	15.9	5.9	7.6	5.5	3.6	9.0	4.8	22.7	8.5	2.1	2.6	4.1	3.8	5.7	7.8	9.9							
3.12.45	32.2	22.2	9.2	28.0	16.7	7.2	18.6	3.8	9.6	4.5	4.0	7.1	21.9	32.2	35.3	7.0	23.4	11.8						
3.12.47a	19.6	11.1	8.3	9.9	4.1	16.2	34.2	3.0	6.8	4.2	3.8	4.7	14.7	40.7	22.0	11.0	28.3	9.7	33.1					
3.12.47b	10.6	12.4	2.4	9.8	4.8	8.2	21.3	6.3	15.8	14.5	2.4	2.9	6.0	11.1	10.2	7.2	14.0	15.1	17.1	29.4				
3.12.48a	7.8	6.1	5.8	6.5	6.0	11.5	8.7	10.1	9.5	9.5	3.1	3.8	9.0	6.1	7.2	11.1	7.5	4.7	9.2	10.1	8.4			

3.12.48c	5.8	3.7	8.2	6.4	3.9	7.3	6.6	8.6	18.6	11.2	5.1	2.6	9.2	5.6	8.5	7.4	7.6	5.5	8.7	5.4	12.8	12.9	
3.12.7	26.1	13.6	16.7	26.5	14.7	7.2	7.7	1.5	6.1	1.7	11.7	1.6	14.7	13.1	30.4	9.6	14.0	7.3	13.4	8.7	3.8	4.8	3.8
3.12.9	6.4	3.5	24.7	7.7	7.3	17.1	2.8	2.9	2.6	3.8	2.3	1.9	6.4	7.6	10.2	23.9	12.2	3.1	6.1	3.0	3.6	4.2	7.4	10.7

References

Clarke, K.R. & Gorley, R.N. (2006) *PRIMER v6: User Manual/Tutorial*. PRIMER-E, Plymouth.

Neldner, V.J., Niehus, R.E., Wilson, B.A., McDonald, W.J.F., Ford, A.J. & Accad, A. (2017) Vegetation of Queensland. Descriptions of Broad Vegetation Groups. Queensland Herbarium, Department of Science, Information Technology and Innovation

Appendix 5

Appendix 5.1 Significance tests for differences in environmental variables between landscapes

I generated one thousand random data points across each landscape and sampled environmental variables at each point using GIS intersects. Environmental variables were available as modelled ANUCLIM datasets (Xu & Hutchinson, 2013). I tested for differences between landscapes using a Mann-Whitney U test as the data was not normally distributed.

Igneous vs. Tertiary landscape	slope (degrees)	elevation (metres)	mean moisture index (lowest quarter)	precipitation (average annual)	temperature seasonality (Coefficient of Variation %)	temperature (average annual)
P value	<0.0001	<0.0001	<0.0001	0.0212	<0.0001	<0.0001
Exact or approximate P value?	Approximate	Approximate	Approximate	Approximate	Approximate	Approximate
One- or two-tailed P value?	Two-tailed	Two-tailed	Two-tailed	Two-tailed	Two-tailed	Two-tailed
Sum of ranks in column A, B	652015, 1174902	578439, 1248478	701660, 1125256	939824, 887093	812149, 1014767	1266767, 560149
Mann-Whitney U	196480	122904	246125	428690	356614	101746
Difference between medians						
Median of column A	1.103, n=954	80, n=954	2, n=954	1382, n=954	67, n=954	25.5, n=954
Median of column B	2.448, n=957	198, n=957	3, n=957	1325, n=957	70, n=957	24.7, n=957
Difference: Actual	1.345	118	1	-57	3	-0.8
Difference: Hodges-Lehmann	1.363	113	1	-23	4	-1

References

Xu, T., & Hutchinson, M. F. (2013). New developments and applications in the ANUCLIM spatial climatic and bioclimatic modelling package. *Environmental Modelling & Software, 40*, 267-279. doi:10.1016/j.envsoft.2012.10.003

Appendix 5.2 Sharpness and Uniqueness values for communities recognised by both methods

Table A5.2.1: Sharpness values for communities recognised by supervised and un-supervised methods on each landscape.

Tertiary landscape		Igneous landscape	
supervised	un-supervised	supervised	un-supervised
2.5	30.0	5.18	81.77
2.7	49.1	34.78	101.19
3.9	66.4	60.97	122.28
36.2	83.9	64.39	147.3
41.5	87.9	71.73	154.61
45.5	97.4	80.88	157.24
45.5	100.1	87.18	158.01
59.1	113.0	88.41	165.38
60.0	117.3	89.33	185.63
60.8	153.6	99.16	191.53
61.1	154.5	101.19	197.13
64.2	202.6	104.79	206.54
74.6	222.1	109.05	215.81
77.6	252.1	115.3	226.02
80.5	268.2	127.15	229.56
81.5	272.3	132.4	259.87
85.4	284.9	133.79	262.96
95.7	336.8	134.12	270.54
96.9		146.71	281.62
99.0		157.57	285.29
99.6		161.19	307.28
101.3		173.14	396.18
101.5		180.13	409.32
107.0		198.55	421.82
116.2		207.03	
123.8		207.86	
126.0		225.92	
126.3		245.5	
130.8		248.42	

	134.0		249.03	
	137.1		253.9	
	144.1		254.5	
	199.8		273.52	
	234.9		282.37	
	247.3		330.1	
	309.4		366.46	
	338.2		387.48	
Average	*106.8*	*160.7*	*167.3*	*226.5*
Standard deviation	*76*	*92.6*	*92.4*	*91.9*

Table A5.2.2: Uniqueness values for communities recognised by supervised and un-supervised methods on each landscape. Communities with no characterising species have an undefined Uniqueness and were given a value of 0 for the purposes of this study

	Tertiary landscape		Igneous landscape	
	supervised	un-supervised	supervised	un-supervised
	Undefined (0)	0.22	0.00	0.41
	Undefined (0)	0.23	0.09	0.51
	Undefined (0)	0.34	0.13	0.51
	0.15	0.39	0.27	0.56
	0.20	0.39	0.28	0.57
	0.21	0.56	0.29	0.66
	0.22	0.58	0.33	0.73
	0.23	0.61	0.35	0.81
	0.24	0.68	0.37	0.85
	0.25	0.72	0.38	0.86
	0.29	0.78	0.39	0.88
	0.31	0.8	0.40	0.89
	0.33	0.93	0.40	0.93
	0.37	0.94	0.45	0.93
	0.37	0.94	0.47	0.94
	0.38	0.97	0.47	0.94
	0.39	1.00	0.50	1.00
	0.40	1.00	0.58	1.00
	0.41		0.60	1.00

	0.41		0.62	1.00
	0.42		0.63	1.00
	0.42		0.64	1.00
	0.44		0.66	1.00
	0.46		0.68	1.00
	0.47		0.70	
	0.51		0.81	
	0.55		0.81	
	0.60		0.83	
	0.62		0.90	
	0.63		0.92	
	0.75		0.95	
	0.75		0.97	
	0.78		1.00	
	0.81		1.00	
	0.94		1.00	
	1.00		1.00	
	1.00		1.00	
Average	*0.44*	*0.67*	*0.59*	*0.83*
Standard deviation	*0.27*	*0.27*	*0.29*	*0.19*

Appendix 5.3 Variability in the similarity of sites within communities in each landscape

Table 5.3.1 Descriptive statistics of the average similarity of sites within communities recognised by each method.

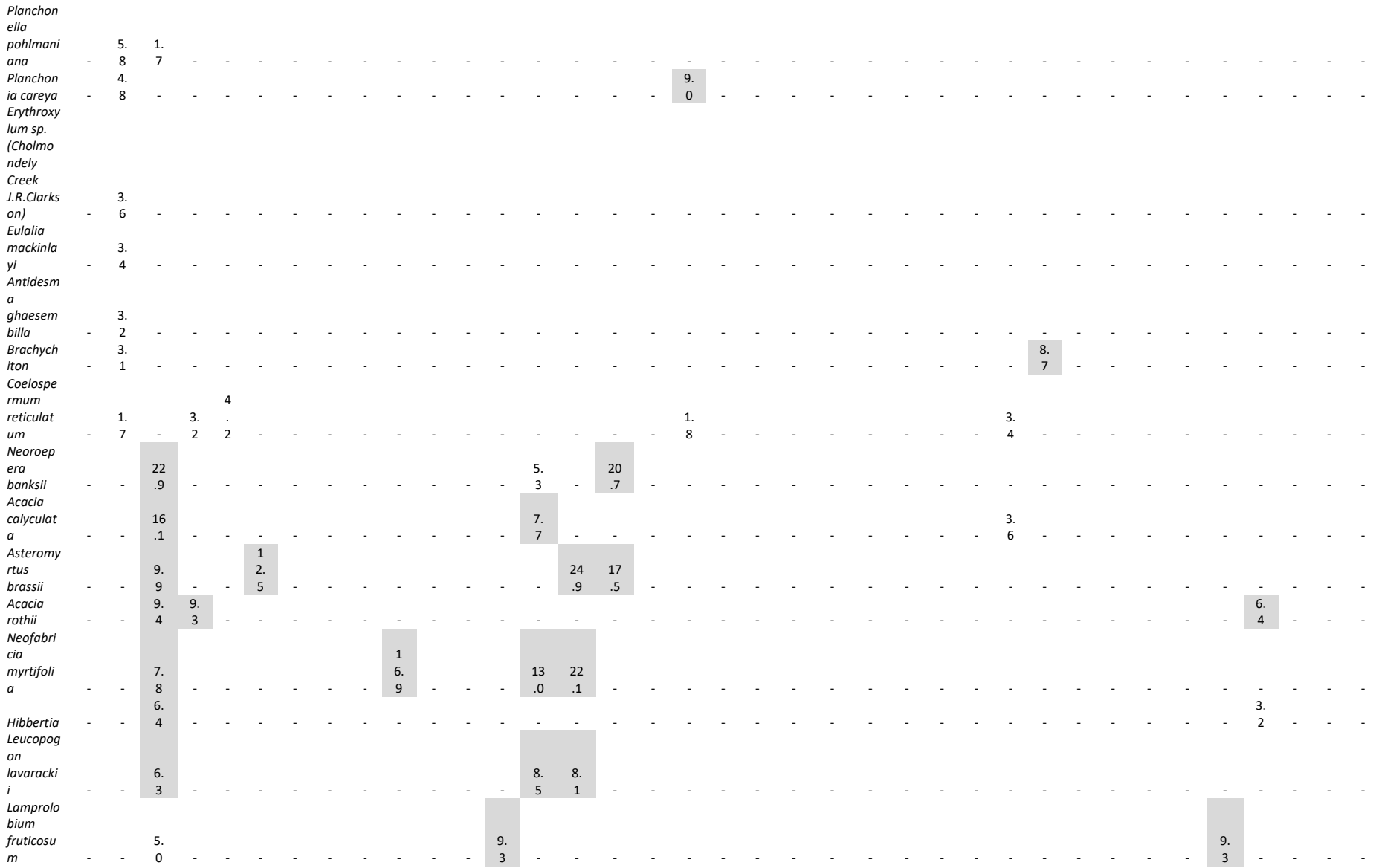
	Tertiary landscape *(n=192)*		Igneous landscape *(n=96)*	
	supervised	un-supervised	supervised	un-supervised
Minimum	26	30	18	18
25th percentile	41	40	27	32
Median	48	47	37	39
75th percentile	53	54	49	45
Maximum	71	61	77	77
Mean	47	47	39	40
Standard deviation	10.7	9.1	16.0	12.2

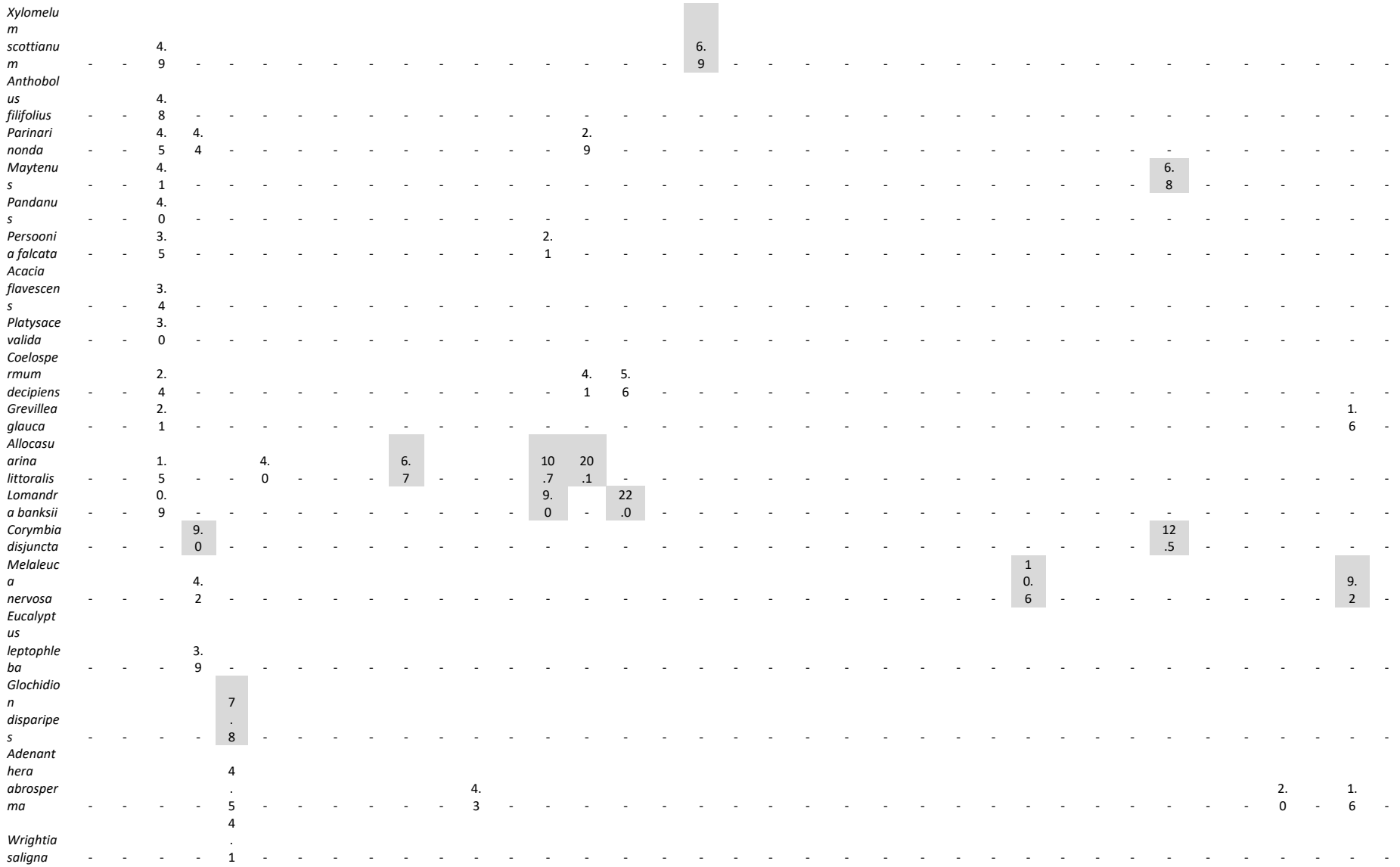
Appendix 5.4 Synoptic tables for the supervised and un-supervised classifications on the landscapes in my study area

In all classifications, groups were standardised to equal size. Characterising species (highlighted) were statistically significantly associated species with a percentage phi-coefficient >6 (to match the threshold used in the un-supervised classification Addicott et al. (2018)). Species in the dataset which are not statistically significantly associated with any community have been removed from the table.

Table A5.4.1. Synoptic table for the supervised classification of the Tertiary landscape.

Community	s1	s10	s10x	s11	s12	s13	s14a	s14c	s15b	s16	s17a	s18	s19	s19x1	s19x2	s19x3	s19x7	s21	s22	s22c	s22x1	s22x2	s23x1	s24a	s25b	s25a	s26	s27a	s27c	s27d	s27x2a	s27x2b	s27x2c	s28a	s28c	s29a	s29d
Species No. sites	3	25	13	14	5	2	3	1	1	3	1	2	1	12	8	2	1	1	1	3	1	7	2	2	1	4	4	1	1	2	8	1	1	5	8	3	8
Capparis	10.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.2	-	-	-	-
Alphitonia pomaderroides	10.1	16	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.3	-
Helicteres prostrate	8.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Erythroxylum ellipticum	8.6	-	-	-	9.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.5	-	-	-	-	-	-	2.3	-
Tinospora smilacina	7.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Indigofera	6.0	-	-	-	5.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.9	-	-	-	-	-	-	-	-
Scleria	0.7	-	-	-	-	-	-	-	47.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia nesophila	-	17.6	15.2	11.1	-	-	-	-	-	-	-	-	-	-	-	-	-	19.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.6
Eucalyptus tetrodonta	-	10.8	5.5	7.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.3	-
Erythrophleum chlorostachys	-	6.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11.9	-


Planchonella pohlmaniana
Planchonia careya
Erythroxylum sp. (Cholmondely Creek J.R.Clarkson)
Eulalia mackinlayi
Antidesma ghaesembilla
Brachychiton
Coelospermum reticulatum
Neoroepera banksii
Acacia calyculata
Asteromyrtus brassii
Acacia rothii
Neofabricia myrtifolia
Hibbertia
Leucopogon lavarackii
Lamprolobium fruticosum


Xylomelum scottianum 4.9 6.9
Anthobolus filifolius 4.8
Parinari nonda 4.5 4.4 2.9
Maytenus 4.1 6.8
Pandanus 4.0
Persoonia falcata 3.5 2.1
Acacia flavescens 3.4
Platysace valida 3.0
Coelospermum decipiens 2.4 4.1 5.6
Grevillea glauca 2.1 1.6
Allocasuarina littoralis 1.5 4.0 6.7 10.7 20.1
Lomandra banksii 0.9 9.0 22.0
Corymbia disjuncta 9.0 12.5
Melaleuca nervosa 4.2 10.6 9.2
Eucalyptus leptophleba 3.9
Glochidion disparipes 7.8
Adenanthera abrosperma 4.5 4.3 2.0 1.6
Wrightia saligna 54.1

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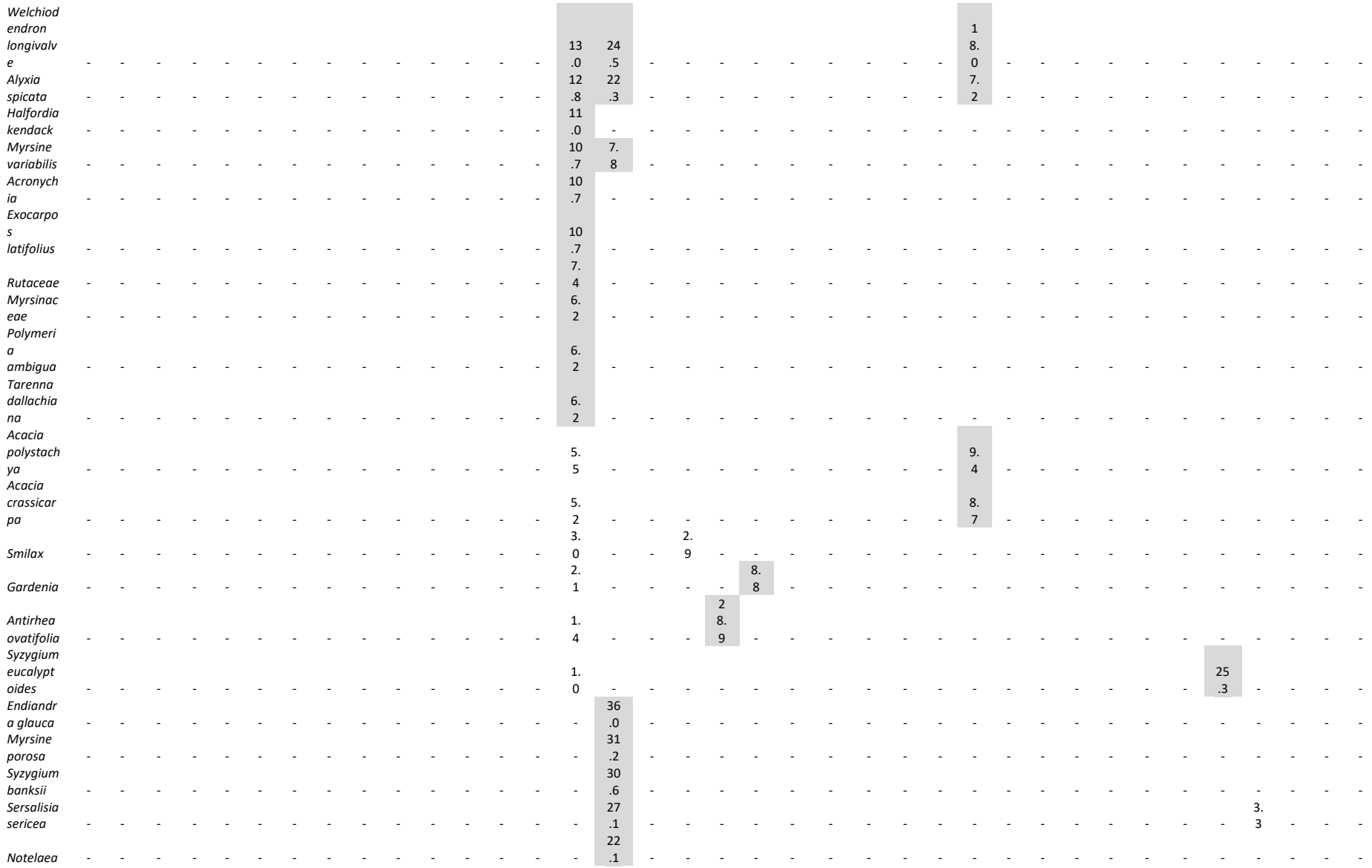
Welchiodendron longivalve 13.0 24.5 18.0
Alyxia spicata 12.8 22.3 7.2
Halfordia kendack 11.0
Myrsine variabilis 10.7 7.8
Acronychia 10.7
Exocarpos latifolius 10.7
Rutaceae 7.4
Myrsinaceae 6.2
Polymeria ambigua 6.2
Tarenna dallachiana 6.2
Acacia polystachya 5.5 9.4
Acacia crassicarpa 5.2 8.7
Smilax 3.0 2.9
Gardenia 2.1 8.8
Antirhea ovatifolia 1.4 28.9
Syzygium eucalyptoides 1.0 25.3
Endiandra glauca 36.0
Myrsine porosa 31.2
Syzygium banksii 30.6
Sersalisia sericea 27.1 3.3
Notelaea 22.1

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Neofabricia myrtifolia	-	21	-	-	15.3	-	-	-	-	-	-	-	-	-	31.8	-	-	-
Gompholobium nitidum	-	19	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Leucopogon ruscifolius	-	19	-	-	-	-	-	-	-	-	-	-	-	-	13.5	-	-	-
Choriceras tricorne	-	19	-	-	-	-	-	-	-	-	-	-	-	-	19.4	-	-	-
Thryptomene oligandra	-	18	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Boronia alulata	-	17	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Jacksonia thesioides	-	16	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Banksia dentata	-	15	-	-	6.5	-	-	-	-	-	-	-	-	-	8.2	-	-	-
Lomandra banksii	-	14	-	-	2.8	-	-	-	-	-	-	-	-	-	6.6	-	-	-
Grevillea pteridifolia	-	12	-	-	3.4	-	-	-	-	-	-	-	-	-	-	-	-	-
Neoroepera banksii	-	10	-	-	24.8	-	-	-	-	-	-	-	-	-	-	-	-	-
Leucopogon lavarackii	-	9.5	-	-	7.5	-	-	-	-	-	-	-	-	-	5.9	-	-	-
Acacia calyculata	-	8.9	-	-	14.6	-	-	-	-	-	-	-	-	-	-	-	4.3	-
Leucopogon yorkensis	-	7.3	-	-	-	-	-	-	-	-	-	-	-	-	31.8	-	-	-
Lithomyrtus obtusa	-	7.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia brassii	-	6.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lithomyrtus retusa	-	6.6	-	-	-	-	-	-	-	-	-	-	-	-	6.6	-	-	-
Melaleuca arcana	-	6.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Gompholobium pinnatum	-	5.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Comesperma secundum	-	4.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Eucalyptus chlorophylla	-	-	22.1	-	-	-	-	-	-	-	-	27	-	-	-	-	-	-
Hakea persiehana	-	-	16.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cayratia	-	-	16.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eucalyptus platyphylla	-	-	13.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia nesophila	-	-	-	30	11.1	22	27	-	-	-	-	-	-	-	-	-	-	-
Acacia simsii	-	-	-	21	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Jagera	-	-	-	13	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Tephrosia	-	-	-	12	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia leptocarpa	-	-	-	9.6	-	-	-	-	-	-	-	-	-	-	6.3	-	-	-
Dolichandrone heterophylla	-	-	-	9.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Terminalia sericocarpa	-	-	-	8.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Buchanania arborescens	-	-	-	8.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Asteromyrtus brassii	-	-	-	-	16.4	-	-	-	-	-	-	-	-	-	33.8	-	-	-
Acacia rothii	-	-	-	-	10.8	9.3	-	-	-	-	-	-	-	-	-	-	-	-
Hibbertia	-	-	-	-	9.2	2.4	-	-	-	-	-	-	-	-	-	-	-	-
Anthobolus filifolius	-	-	-	-	8.7	-	-	-	-	-	-	-	-	-	-	-	-	-
Lamprolobium fruticosum	-	-	-	-	8.6	-	-	-	-	-	-	-	-	-	-	-	-	-
Eucalyptus tetrodonta	-	-	-	-	8.1	14	-	13	-	-	13	-	-	-	-	-	-	-
Parinari nonda	-	-	-	-	7	4.1	-	-	-	-	-	-	-	-	2.1	-	-	-
Xylomelum scottianum	-	-	-	-	6.5	2.4	11	-	-	-	2	-	-	-	-	-	-	-
Persoonia falcata	-	-	-	-	6	3.2	-	-	-	-	-	-	-	-	-	-	-	2.2
Grevillea glauca	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-	-	-	-
Platysace valida	-	-	-	-	4.2	-	-	-	-	-	-	-	-	-	-	-	-	-

Pandanus	-	-	-	-	4.2	-	-	-	-	-	-	-	-	-	-	-	-	-
Clerodendrum	-	-	-	-	3.3	-	-	-	-	-	-	-	-	-	-	-	-	-
Coelospermum decipiens	-	-	-	-	3.2	-	8.2	-	-	-	-	-	-	-	3.7	-	-	-
Xanthorrhoea johnsonii	-	-	-	-	2.9	-	-	-	-	-	-	-	-	-	-	-	-	-
Planchonella pohlmaniana	-	-	-	-	2.3	6.6	-	-	-	-	-	-	-	-	-	-	-	-
Acacia flavescens	-	-	-	-	0.8	3.3	-	-	-	-	-	-	-	-	-	-	-	-
Alyxia spicata	-	-	-	-	0.8	-	-	-	-	-	-	-	-	-	15.8	7.7	-	-
Erythrophleum chlorostachys	-	-	-	-	-	9.9	-	6.2	-	-	9.3	-	-	-	-	-	-	21
Helicteres semiglabra	-	-	-	-	-	9.7	-	-	-	-	-	-	-	-	-	-	-	-
Cycas yorkiana	-	-	-	-	-	7.2	-	-	-	-	-	-	-	-	-	-	-	-
Erythroxylum sp. (Cholmondely Creek J.R.Clarkson 9	-	-	-	-	-	6.1	-	-	-	-	-	-	-	-	-	-	-	-
Planchonia careya	-	-	-	-	-	4.8	14	-	-	-	-	-	-	-	-	-	-	-
Brachychiton	-	-	-	-	-	4.8	-	-	-	-	4.7	-	-	-	-	-	-	-
Decaschistia peninsularis	-	-	-	-	-	4.2	-	3.8	-	-	-	-	-	-	-	-	-	-
Coelospermum reticulatum	-	-	-	-	-	4.2	-	-	-	-	-	-	-	-	-	-	5.4	-
Corymbia disjuncta	-	-	-	-	-	3.5	-	-	-	-	6.6	-	-	-	-	-	-	-
Brachychiton muellerianus	-	-	-	-	-	2.4	-	-	-	-	-	-	-	-	-	-	-	-
Alphitonia pomaderroides	-	-	-	-	-	1.9	-	-	-	-	3	-	-	-	-	-	-	5.7
Livistona muelleri	-	-	-	-	-	-	11	-	-	-	-	-	-	-	-	12.4	-	-
Siphonodon	-	-	-	-	-	-	9.3	-	-	-	-	-	-	-	-	-	-	-
Grewia retusifolia	-	-	-	-	-	-	7.1	3.5	-	-	-	-	-	-	-	-	-	-

Grewia latifolia	-	-	-	-	-	-	5.6	-	-	-	-	-	-	-	-	-	-	-
Smilax	-	-	-	-	-	-	3.6	-	-	-	-	-	-	-	5.8	-	-	-
Corymbia stockeri	-	-	-	-	-	-	-	19	11.8	15	-	-	-	-	-	-	-	8.5
Bossiaea armitii	-	-	-	-	-	-	-	8.5	-	-	-	-	-	-	-	-	-	-
Adenanthera abrosperma	-	-	-	-	-	-	-	5	-	-	-	-	-	-	-	-	-	-
Sersalisia unmackiana	-	-	-	-	-	-	-	4.4	-	-	-	-	-	-	-	-	-	-
Neofabricia sericisepala	-	-	-	-	-	-	-	2.3	-	-	-	-	-	-	-	-	-	24
Eucalyptus cullenii	-	-	-	-	-	-	-	-	46.5	-	-	-	-	-	-	-	-	-
Melaleuca stenostachya	-	-	-	-	-	-	-	-	3.9	18	-	35	-	-	-	-	-	-
Petalostigma banksii	-	-	-	-	-	-	-	-	1.1	7.7	-	-	-	-	-	-	-	6.3
Acacia ommatosperma	-	-	-	-	-	-	-	-	-	6.5	-	-	-	-	-	-	-	-
Melaleuca viridiflora	-	-	-	-	-	-	-	-	-	5.2	-	6.5	13	-	-	-	-	-
Corymbia clarksoniana	-	-	-	-	-	-	-	-	-	-	8.4	-	24	-	-	-	-	-
Indigofera	-	-	-	-	-	-	-	-	-	-	7.9	-	-	-	-	-	-	-
Atalaya variifolia	-	-	-	-	-	-	-	-	-	-	7.4	-	-	-	-	-	-	-
Erythroxylum ellipticum	-	-	-	-	-	-	-	-	-	-	6.4	-	-	-	-	-	-	6.1
Waltheria indica	-	-	-	-	-	-	-	-	-	-	4.8	-	-	-	-	-	-	-
Tinospora smilacina	-	-	-	-	-	-	-	-	-	-	3.3	-	-	-	-	-	-	-
Croton arnhemicus	-	-	-	-	-	-	-	-	-	-	3.2	-	-	-	-	-	-	-
Denhamia	-	-	-	-	-	-	-	-	-	-	1.7	-	-	-	-	-	-	-
Sauropus	-	-	-	-	-	-	-	-	-	-	1.6	-	-	-	-	-	-	-
Wrightia saligna	-	-	-	-	-	-	-	-	-	-	1.4	-	-	-	-	-	-	-
Corymbia hylandii	-	-	-	-	-	-	-	-	-	-	-	20	-	-	-	-	-	-

Melaleuca foliolosa	-	-	-	-	-	-	-	-	-	-	-	9.2	-	-	-	-	-	-
Acacia holosericea	-	-	-	-	-	-	-	-	-	-	-	8.1	-	-	-	-	-	-
Acacia leiocalyx	-	-	-	-	-	-	-	-	-	-	-	7.7	-	-	-	-	-	-
Dodonaea viscosa	-	-	-	-	-	-	-	-	-	-	-	7.7	-	-	-	-	-	-
Harrisonia brownii	-	-	-	-	-	-	-	-	-	-	-	7.7	-	-	-	-	-	-
Acacia leptostachya	-	-	-	-	-	-	-	-	-	-	-	7.7	-	-	-	-	-	-
Terminalia platyptera	-	-	-	-	-	-	-	-	-	-	-	-	12	-	-	-	-	-
Flueggea virosa subsp. melanthesoides	-	-	-	-	-	-	-	-	-	-	-	-	5.4	-	-	-	-	-
Amyema villiflora	-	-	-	-	-	-	-	-	-	-	-	-	-	17.3	-	-	-	-
Alphitonia excelsa	-	-	-	-	-	-	-	-	-	-	-	-	-	15.2	-	-	3	-
Callitris intratropica	-	-	-	-	-	-	-	-	-	-	-	-	-	-	30.4	-	-	-
Dodonaea polyandra	-	-	-	-	-	-	-	-	-	-	-	-	-	-	22.7	-	-	-
Welchiodendron longivalve	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.9	23.3	-	-
Olax pendula	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.5	-	-	-
Deplanchea tetraphylla	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.9	-	-	-
Endiandra glauca	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.3	-	-	-
Syzygium banksii	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11.8	-	-	-
Myrsine variabilis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11.7	-	-	-
Exocarpos latifolius	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.9	-	-	-
Atractocarpus sessilis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.7	-	-	-
Acacia polystachya	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.8	25.1	-	-
Acacia crassicarpa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.6	19.2	-	-

Acronychia	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8	-	-	-
Myrmecodia	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.5	-	-	-
Syzygium eucalyptoides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3.9	-	-	-
Gardenia	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.7	29.9	-	-
Antirhea ovatifolia	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.9	13	-	-
Corymbia novoguinensis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	57.9	-	-
Corymbia tessellaris	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	33.1	-	-
Bridelia tomentosa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	20.8	-	-
Terminalia muelleri	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.7	-	-
Tabernaemontana orientalis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.3	-	-
Mallotus nesophilus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	16.5	-	-
Vandasina retusa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.9	-	-
Cycas media	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.9	-	-
Drypetes deplanchei	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.4	-	-
Cupaniopsis anacardioides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.4	-	-
Diospyros hebecarpa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.6	-	-
Abrus precatorius	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.6	-	-
Zanthoxylum rhetsa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.6	-	-
Rhodomyrtus macrocarpa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.6	-	-
Pleurostylia opposita	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.6	-	-

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Flueggea virosa subsp. melanthesoides	-	5.3	-	16.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia stockeri	-	-	54.2	-	-	-	-	-	-	-	-	-	-	-	7.5	-	-	-	-	-	-	-	-	-	-	-
Dalbergia densa var. australis	-	-	23.7	-	-	-	-	-	-	-	-	-	-	-	-	-	3.2	-	-	-	-	-	-	-	-	-
Welchiodendron longivalve	-	-	15.7	-	-	-	-	-	-	-	-	-	-	-	-	-	34.7	-	-	-	-	-	-	-	-	-
Antirhea ovatifolia	-	-	15.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Smilax	-	-	13.3	-	-	-	-	-	-	-	-	-	9.4	-	-	-	7.3	-	-	-	-	-	-	-	-	-
Livistona muelleri	-	-	12.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia leptocarpa	-	-	9.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia polystachya	-	-	0.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eucalyptus leptophleba	-	-	-	50.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Timonius timon	-	-	-	7.9	-	-	-	-	-	-	-	-	4.6	-	-	-	-	-	-	-	-	-	-	-	-	-
Ficus opposita	-	-	-	7.7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Grewia latifolia	-	-	-	6.7	6.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mallotus philippensis	-	-	-	5.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia disjuncta	-	-	-	-	53.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Piliostigma malabaricum	-	-	-	-	21.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Caladenia	-	-	-	-	-	31.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Randia	-	-	-	-	-	17.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rapanea	-	-	-	-	-	17.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rubiaceae	-	-	-	-	-	14.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Dendrobium discolor	-	-	-	-	-	13.9	-	-	-	-	6.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Parsonsia	-	-	-	-	-	10.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lithomyrtus obtusa	-	-	-	-	-	8.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Leptospermum purpurascens	-	-	-	-	-	-	64.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Shonia tristigma	-	-	-	-	-	-	20.1	-	-	-	-	-	-	-	-	-	-	-	-	10.6	-	-	-	-	-	-
Hovea	-	-	-	-	-	-	19.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Labichea nitida	-	-	-	-	-	-	15.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lithomyrtus retusa	-	-	-	-	-	-	12.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Leucopogon yorkensis	-	-	-	-	-	-	9.7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Grevillea pteridifolia	-	-	-	-	-	-	8.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia midgleyi	-	-	-	-	-	-	5.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Imperata cylindrica	-	-	-	-	-	-	-	84.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Heteropogon contortus	-	-	-	-	-	-	-	61.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.0	-	-
Mnesithea rottboellioides	-	-	-	-	-	-	-	56.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cymbopogon refractus	-	-	-	-	-	-	-	44.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rottboellia cochinchinensis	-	-	-	-	-	-	-	40.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Commelina	-	-	-	-	-	-	-	11.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eragrostis	-	-	-	-	-	-	-	10.4	-	-	-	-	-	-	-	-	-	-	-	-	-	15.8	-	-	-
Evolvulus alsinoides	-	-	-	-	-	-	-	9.1	-	-	-	-	-	-	-	-	-	-	-	-	-	10.0	9.1	-	-
Ectrosia leporina	-	-	-	-	-	-	-	-	47.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eriachne	-	-	-	-	-	-	-	-	38.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rhynchosia	-	-	-	-	-	-	-	-	34.9	3.9	-	-	-	-	-	-	-	-	-	-	-	2.9	-	-	-
Polycarpaea spirostylis	-	-	-	-	-	-	-	-	31.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Scleria	-	-	-	-	-	-	-	-	30.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Schizachyrium	-	-	-	-	-	-	-	-	30.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Fimbristylis	-	-	-	-	-	-	-	-	27.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pseudopogonatherum contortum	-	-	-	-	-	-	-	-	21.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ptilotus	-	-	-	-	-	-	-	-	21.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Atalaya hemiglauca	-	-	-	-	-	-	-	-	17.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Tephrosia	-	-	-	-	-	-	-	-	15.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mesosphaerum suaveolens	-	-	-	-	-	-	-	-	12.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia papuana	-	-	-	-	-	-	-	-	12.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Sersalisia unmackiana	-	-	-	-	-	-	-	-	12.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia hemignosta	-	-	-	-	-	-	-	-	11.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Aristida	-	-	-	-	-	-	-	-	6.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Heteropogon triticeus	-	-	-	-	-	-	-	-	0.4	-	-	-	-	-	-	-	-	-	-	-	-	72.1	-	-	-
Rock pavement	-	-	-	-	-	-	-	-	-	99.7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Acacia disparrima subsp. calidestris	-	-	-	-	-	-	-	-	-	-	38.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Eucalyptus crebra	-	-	-	-	-	-	-	-	-	-	35.7	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Manilkara kauki	-	-	-	-	-	-	-	-	-	-	33.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Diospyros	-	-	-	-	-	-	-	-	-	-	32.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Terminalia subacroptera	-	-	-	-	-	-	-	-	-	-	30.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Micromelum minutum	-	-	-	-	-	-	-	-	-	-	28.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bombax ceiba var. leiocarpum	-	-	-	-	-	-	-	-	-	-	26.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cupaniopsis fleckeri	-	-	-	-	-	-	-	-	-	-	26.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Argophyllum verae	-	-	-	-	-	-	-	-	-	-	26.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Alphitonia excelsa	-	-	-	-	-	-	-	-	-	-	26.1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Psychotria	-	-	-	-	-	-	-	-	-	-	19.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Convolvulaceae	-	-	-	-	-	-	-	-	-	-	12.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Dodonaea viscosa	-	-	-	-	-	-	-	-	-	-	12.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Chionanthus ramiflorus	-	-	-	-	-	-	-	-	-	-	11.2	-	-	-	-	-	-	-	-	-	-	-	-	-	40.1	-
Dioscorea	-	-	-	-	-	-	-	-	-	-	10.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Abrus precatorius	-	-	-	-	-	-	-	-	-	-	10.2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Endiandra glauca	-	-	-	-	-	-	-	-	-	-	-	69.7	-	1.3	-	-	-	-	-	-	-	-	-	-	-	-
Myrsine variabilis	-	-	-	-	-	-	-	-	-	-	-	46.5	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Litsea breviumbellata	-	-	-	-	-	-	-	-	-	-	-	40.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rhodamnia	-	-	-	-	-	-	-	-	-	-	-	37.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Muehlenbeckia zippelii	-	-	-	-	-	-	-	-	-	-	-	37.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cyclophyllum coprosmoides	-	-	-	-	-	-	-	-	-	-	-	37.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Guioa acutifolia	-	-	-	-	-	-	-	-	-	-	-	31.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mischocarpus	-	-	-	-	-	-	-	-	-	-	-	31.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cryptocarya vulgaris	-	-	-	-	-	-	-	-	-	-	-	31.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Stephania japonica	-	-	-	-	-	-	-	-	-	-	-	17.4	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Flagellaria indica	-	-	-	-	-	-	-	-	-	-	-	16.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Corymbia nesophila	-	-	-	-	-	-	-	-	-	-	-	-	62.2	18.2	-	-	-	-	-	-	-	-	-	-	-	-
Eucalyptus tetrodonta	-	-	-	-	-	-	-	-	-	-	-	-	24.1	22.1	32.5	-	-	-	-	-	-	-	-	-	-	-

Coelospermum reticulatum	-	-	-	-	-	-	-	-	-	-	-	-	16.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Acacia flavescens	-	-	-	-	-	-	-	-	-	-	-	-	15.2	-	9.4	-	-	-	-	-	-	-	-	-	-	-	-
Leucopogon ruscifolius	-	-	-	-	-	-	-	-	-	-	-	-	11.3	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Parinari nonda	-	-	-	-	-	-	-	-	-	-	-	-	7.8	-	5.0	-	-	-	-	-	-	-	-	-	-	-	-
Persoonia falcata	-	-	-	-	-	-	-	-	-	-	-	-	7.7	7.3	-	-	-	-	-	-	-	-	-	-	-	-	-
Eulalia mackinlayi	-	-	-	-	-	-	-	-	-	-	-	-	3.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Lophostemon suaveolens	-	-	-	-	-	-	-	-	-	-	-	-	1.8	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Asteromyrtus brassii	-	-	-	-	-	-	-	-	-	-	-	-	-	34.7	-	-	-	-	-	12.2	-	-	-	-	-	-	-
Acacia rothii	-	-	-	-	-	-	-	-	-	-	-	-	-	30.3	-	-	-	-	2.1	-	-	-	-	-	-	-	-
Grevillea glauca	-	-	-	-	-	-	-	-	-	-	-	-	-	21.4	4.0	-	-	-	-	-	-	-	-	-	-	-	-
Neofabricia myrtifolia	-	-	-	-	-	-	-	-	-	-	-	-	-	20.6	-	-	-	-	-	16.3	-	-	-	-	-	-	-
Xanthorrhoea johnsonii	-	-	-	-	-	-	-	-	-	-	-	-	-	17.9	-	-	-	-	10.3	18.9	-	-	-	-	-	-	-
Hibbertia banksii	-	-	-	-	-	-	-	-	-	-	-	-	-	17.8	-	-	-	-	-	7.3	-	-	-	-	-	-	-
Allocasuarina littoralis	-	-	-	-	-	-	-	-	-	-	-	-	-	17.3	-	-	-	-	-	31.4	-	-	-	-	-	-	-
Banksia dentata	-	-	-	-	-	-	-	-	-	-	-	-	-	15.2	-	-	-	-	-	-	-	-	-	-	-	-	-
Helicteres semiglabra	-	-	-	-	-	-	-	-	-	-	-	-	-	13.8	-	-	12.2	-	-	-	-	-	-	-	-	-	-
Acacia calyculata	-	-	-	-	-	-	-	-	-	-	-	-	-	13.3	-	-	-	-	-	14.7	-	-	-	-	-	-	-
Choriceras tricorne	-	-	-	-	-	-	-	-	-	-	-	-	-	10.7	-	-	-	-	-	24.0	-	-	-	-	-	-	-
Melaleuca viridiflora	-	-	-	-	-	-	-	-	-	-	-	-	-	9.1	-	-	-	-	25.5	-	-	-	-	-	-	-	-
Coelospermum decipiens	-	-	-	-	-	-	-	-	-	-	-	-	-	4.3	3.4	-	-	-	-	-	-	-	-	-	-	-	-
Grevillea parallela	-	-	-	-	-	-	-	-	-	-	-	-	-	4.3	-	-	-	-	-	-	-	-	-	-	-	-	-
Hibbertia	-	-	-	-	-	-	-	-	-	-	-	-	-	1.9	-	-	-	-	-	12.6	-	-	-	-	-	-	-
Acacia leptostachya	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.6	-	-	-	-	-	-	-	-	-	-	-	-
Xylomelum scottianum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.6	-	-	-	-	-	-	-	-	-	-	-	-
Petalostigma pubescens	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5.6	-	-	-	-	-	-	-	-	-	-	-	-
Planchonella pohlmaniana	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.8	-	-	-	-	-	-	-	-	-	-	-	-
Croton arnhemicus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	27.6	-	-	-	-	-	-	-	-	-	-	-
Syzygium suborbiculare	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	27.1	-	-	-	-	-	-	-	-	-	-	-
Acacia brassii	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	32.4	-	-	-	-	-	-	-	-	-	-
Cochlospermum gillivraei	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.1	-	-	-	-	-	-	-	-	-	-

Ampelocissus acetosa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11.3	-	-	-	-	-	-	-	-
Carissa lanceolata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.0	-	-	-	-	-	-	-	-
Euroschinus falcatus	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5.6	-	-	-	-	-	-	13.1	-
Premna	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4.9	-	-	-	-	-	-	-	-
Santalum lanceolatum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.4	-	-	-	-	-	-	-	-
Melaleuca citrolens	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	48.9	-	-	-	-	-	-	-
Melaleuca foliolosa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	25.3	-	-	-	-	-	-	-
Petalostigma banksii	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	21.1	-	-	-	-	-	-	-
Acacia multisiliqua	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.0	-	-	-	-	-	-	-
Terminalia platyptera	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.1	-	-	-	-	-	-	-
Asteromyrtus lysicephala	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	24.2	-	-	-	-	-
Melaleuca arcana	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	21.9	-	-	-	-	-
Jacksonia thesioides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.0	-	-	-	-	-
Stenanthemum argenteum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.5	-	-	-	-	-
Leucopogon lavarackii	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.3	-	-	-	-	-
Sarga plumosum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	51.8	-	-	-
Ipomoea	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	26.8	-	-	-
Tricoryne anceps	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.9	-	-	-
Cassytha filiformis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8.6	-	-	-
Atalaya variifolia	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	17.4	-	-
Dischidia major	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	16.2	-	-
Dendrobium johannis	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	14.7	-	-
Wollastonia biflora	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.2	-	-
Eucalyptus brassiana	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	47.4	-
Calophyllum sil	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	32.8	-
Wikstroemia indica	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	30.3	-
Celtis paniculata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28.9	-
Polyscias elegans	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28.9	-
Ganophyllum falcatum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28.1	-
Litsea glutinosa	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	28.1	-
Wodyetia bifurcata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	26.1	-

Buchanania arborescens	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	23.7	-
Cupaniopsis anacardioides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	21.9	-
Drypetes deplanchei	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12.3	-
Opilia amentacea	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11.4	-
Notelaea	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.5	-
Alectryon	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	10.5	-
Larsenaikia ochreata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	9.9	-
Tabernaemontana pandacaqui	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	26.1
Tacca leontopetaloides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15.8

Appendix 6

Appendix 6.1 Calcuations used for estimating C storage potential of the mangrove forests and saltmarshes of CYP

Data from plots in my study area to calculate stem density and diameter and canopy cover for comparison with plots used by Donato et al (2011). Stem diameter was calculated from stem density and basal area (BA) measures in plot data. The mangrove forests fall within the groups they refer to as small to intermediate (tree diameter <10 - 20 cm or canopy height <4 - 15 m)

RE identifier	Number of plots	Average height	Height range	Average BA (m^2/ha)	Stem density / ha average	Stem density / ha range	Average stem diameter	Stem diameter range	Average crown cover (%)	Crown cover (%) range
3.1.1a	8	13	5-12	20	2252	500 - 5581	10.7	8 - 18	65	50 - 75
3.1.1b	4	14	10-22	12	not available	not available	not available	not available	56	40 - 75
3.1.1c	0	not available	not available	not available	not available	not available	not available	not available	not available	not available
3.1.2a	3	6.5	5-8	12	1715	633 - 2900	9.4	9 - 9.5	44	27 - 60
3.1.2b	0	not available	not available	not available	not available	not available	not available	not available	not available	not available
3.1.3	11	10	3.5-14	23	4549	533 - 13000	8.0	6- 10	54	31 - 80
3.1.4	3	6.5	5.5-8	13	6020	900 - 15400	5.3	3.6 - 13	57	46 - 75

Figures from plots in Donato et al (2011) which fall in the groups they considered small to intermediate stature forests .

Site	stature *s = small i = intermdiate*	Geomorphic setting	Total ecosystem C (Mg C/ha)	Total above-ground C (Mg C/ha)	Total below-ground C Mg C/ha
G1	s	Estuarine	547	107.4	439.6
G2	i	Estuarine	584.1	60	524.1
J1	s	Estuarine	437	6.5	430.5

J2	s	Estuarine	736.4	17.1	719.4
B1	i	Estuarine	1044.2	78.3	965.9
S6	i	Estuarine	2202.9	111	2091.8
S1	s	Oceanic	734	145.8	588.2
S2	s	Oceanic	415.1	115	300
S3	s	Oceanic	774.6	68	706.6
S4	s	Oceanic	859.3	87.4	772
S5	i	Oceanic	716	97.7	618.2
P1	s	Oceanic	706.5	118.1	588.4
Y1	s	Oceanic	823.2	140	683.2
Y2	i	Oceanic	895.8	205.3	690.5
estuarine mean			**925.3**	**63.4**	**861.9**
estuarine s.e.m			269.6	18.1	259.6
oceanic mean			**740.6**	**122.2**	**618.4**
oceanic s.e.m			269.6	18.1	259.6
total mean			**819.7**	**97.0**	**722.7**
s.e.m			177.3	21.0	174.3

Estimates of C storage in mangrove forests and saltmarshes of CYP. Estimates were calculated by extrapolating figures from Donato et al across the mapped areas on CYP.

	Mapped area (ha)	Above ground C stock (Mg/ha) *(from Donato et al 2011)*	Above ground C stock (Mg)	Below ground C stock (Mg/ha) *(from Donato et al 2011)*	Below ground C stock (Mg)	Total C stock (Mg/ha) *(from Donato et al 2011)*	Total C stock (Mg)
Mangrove forests							
All forests	**153,716**	**97.0**	**14,906.1**	**819.7**	**111,097.1**	**722.7**	**126,001.0**

s.e.m		*21.0*	*3,221.5*	*177.3*	*26,792.1*	*174.3*	*27,248.2*
Estuarine							
3.1.1	68,542		4,345.6		59,076.3		63,421.9
3.1.2	8,382		531.4		7,224.4		7,755.9
3.1.3	51,115		3,240.7		44,056.0		47,296.7
3.1.4	2,271		144.0		1,957.4		2,101.4
Total	**130,310**	**63.4**	**8,261.7**	**861.9**	**112,314.2**	**925.3**	**120,575.8**
s.e.m		*18.1*	*2,357.2*	*259.6*	*33,833.2*	*269.6*	*35,127.7*
Oceanic							
3.1.1	19,448		2,375.8		12,026.4		14,403.2
3.1.2	3,343		408.4		2,067.3		2,475.8
3.1.3	553		67.6		342.0		409.6
3.1.4	62		7.6		38.3		45.9
Total	**23,406**	**122.2**	**2,859.3**	**618.4**	**14,474.0**	**740.6**	**17,334.5**
s.e.m		*15.0*	*351.0*	*50.7*	*1,186.7*	*52.4*	*1,227.1*
Saltmarsh *(from Macreadie et al)*	107,843					77.9	8,403
st dev						*3.35*	*361.3*

References

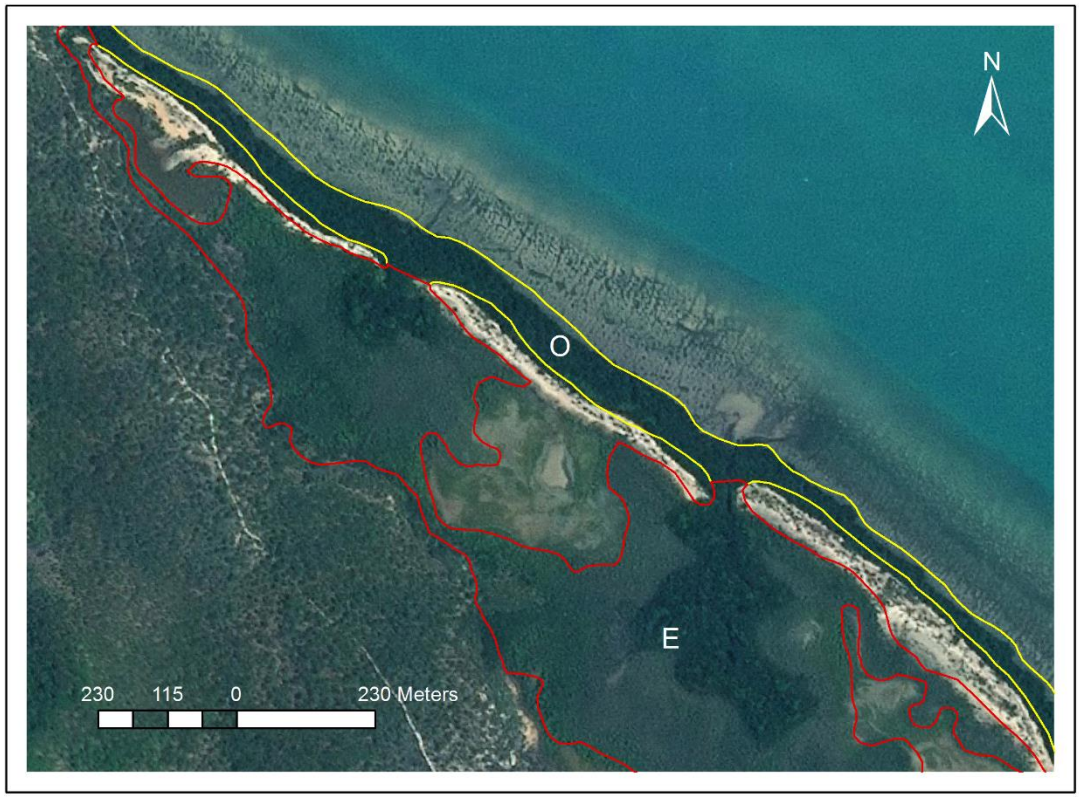
Donato, D. C., Kauffman, J. B., Murdiyarso, D., Kurnianto, S., Stidham, M., & Kanninen, M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience, 4*(5), 293-297. doi:10.1038/ngeo1123

Appendix 6.2 Examples of implementation of guidelines for determining estuarine mangroves and oceanic mangroves using the regional ecosystem mapping of Queensland

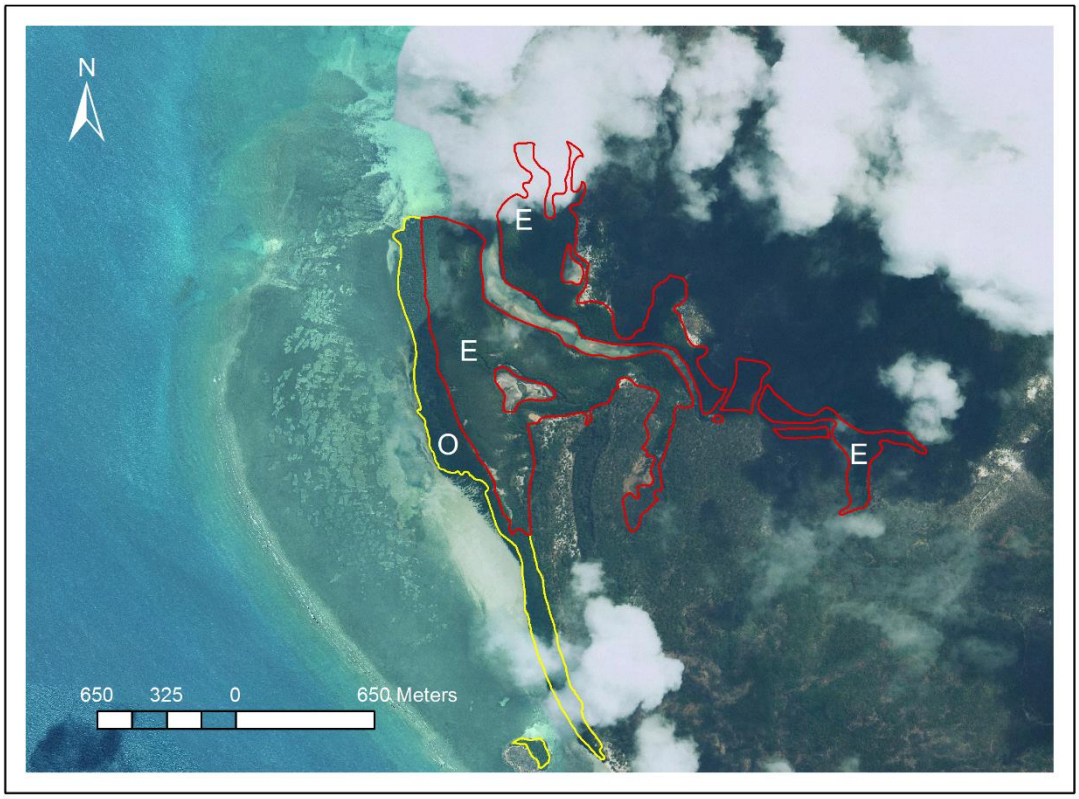
RE mapping is available from

http://qldspatial.information.qld.gov.au/catalogue/custom/index.page).

On figures ‘O’ = oceanic, ‘E’= estuarine mangroves.



Example of guideline 1. Where there is a barrier between estuarine mangroves and mangroves on the coast, the coastal mangroves are mapped as oceanic



Example of guideline 4. Where mangroves from an estuary reach the shoreline AND there is a fringing reef, the boundary of the fringing mangrove is used to define the oceanic mangrove.

Appendix 6.3 Descriptions of the mangrove forest and saltmarsh communities of CYP bioregion

These communities are labelled with an identifier representing the bioregion, the geomorphological category (representing the inter-tidal zone) and the plant community number. The short label description, mapped area and distinguishing community information only is presented here as the full information with detailed descriptions, conservation status and ecological notes are available on-line at (http://www.qld.gov.au/environment/plants-animals/plants/ecosystems/). The identifier and the short label descriptions follow the standard Queensland government naming conventions (Neldner et al 2019).

RE identifier	Short Label Description	Mapped pre-clear area (km^2)	Number of vegetation communities	Derivation	Number of detailed sites	Comments & Distribution
Mangrove forests (59% of land zone, 1 358 km^2)						
3.1.1		**840**	3			
Vegetation communities:						
3.1.1a	*Rhizophora stylosa* +/- *Bruguiera* spp. closed forest. *(majority mapped as 3.1.1)*	180		Quantitative	8	Occurs in downstream estuarine areas in the mid to low intertidal zones throughout the bioregion. Also mapped in the Torres Strait, where it can occur as a *Pemphis acidula* (digging-stick tree) and *Osbornia octodonta* (myrtle mangrove) shrubland. The majority of this vegetation community is mapped as 3.1.1.
3.1.1b	*Bruguiera* spp. closed forest. *(majority mapped as 3.1.1)*	3		Quantitative	4	Occurs in downstream to intermediate estuarine areas in the mid to high intertidal zones. Occurs close association with 3.1.1a and around the extent of the coastline but is not always present. The majority of this vegetation community is mapped as 3.1.1.
3.1.1c	*Heritiera littoralis* +/- *Melaleuca cajuputi* open forest. *(majority mapped as 3.1.1)*	15 ha		Qualitative	0	Where it occurs this community occupies a narrow band on the landward side of more typical mangrove communities where there is some freshwater seepage. *Nypa fruticans* (mangrove palm) also occurs in the tidal reaches of the Wenlock, Ducie, Pascoe and McIvor

						Rivers. The majority of this vegetation community is mapped as 3.1.1.
3.1.2		**260**	2			
Vegetation communities:						
3.1.2a	*Avicennia marina* +/- *Ceriops tagal* low open forest.	260		Quantitative	3	Occurs in downstream to intermediate estuarine areas in the mainly high intertidal zones throughout the bioregion. On Biogu Island can occur as a tall open forest.
3.1.2b	Semi deciduous transitional vine forest + *Xylocarpus granatum* + *Manilkara kauki* on calcareous beach ridges.	6 ha		Qualitative	0	Restricted to Sassie Island in the Torres Strait. Closely associated with open forests of *Avicennia marina* (RE 3.1.2a).
3.1.3	*Ceriops tagal* and/or *C. australis* +/- *Avicennia marina* low open forest.	**415**		Quantitative	12	Occurs in downstream to intermediate estuarine areas in the high to mid intertidal zones throughout the bioregion.
3.1.4	*Excoecaria agallocha* +/- *Aegiceras corniculatum* low open forest.	**25**		Quantitative	3	Occurs in upstream to intermediate estuarine areas in the mainly high intertidal zones throughout the bioregion. Scattered throughout coastal areas including the Torres Strait. Common in Princess Charlotte Bay area.

Salt marshes and salt flats (41% of land zone, 1 064 km^2)

3.1.5	*Sporobolus virginicus* closed tussock grassland.	**133**	Quantitative	3	Occurs on coastal plains in the high intertidal zone subject to infrequent inundation. Largest areas are in Lakefield national park and on west coast, but smaller areas occur around the extent of the coastline and some Torres Strait islands.
3.1.6	Sparse herbland or bare saltpans on salt plains and saline flats.	**921**	Quantitative	8	Occurs across the bioregion in the high intertidal zone subject to infrequent inundation. A range of halophytic forbs may occur in small areas of saltmarsh within this community.
3.1.6x1	*Batis argillicola* low shrubland	**5 ha**	Quantitative	2	Occurs infrequently at the landward edge of mangrove on the margins of salt pans.
3.1.7	*Schoenoplectus subulatus* sedgelands in depressions on tidal flats.	**10 ha**	Qualitative	0	Restricted to Saibai and Boigu islands in Torres Strait. Important estuarine habitat for estuarine crocodile, migratory waders, Jabiru and Sarus Cranes. Cultural resource for traditional fishing and hunting.

Appendix 6.4. Species list of mangroves in Cape York Peninsula bioregion

See manuscript text for method of species compilation. Nomenclature follows Bostock & Holland (2018).

Appendix 6.4. Species list of mangroves in Cape York Peninsula bioregion. See manuscript text method of species compilation. Nomenclature follows Bostock & Holland (2018).

Genus_species_subspecies	Present in CYP	From Tomlinson 2016 t= true mangrove a= mangrove associate
Heritiera littoralis	1	a
Acanthus ebracteatus subsp. ebarbatus	1	a
Acanthus ebracteatus subsp. ebracteatus	1	a
Acanthus ilicifolius	1	a
Dolichandrone spathacea	1	a
Cynometra iripa	1	a
Cynometra ramiflora	1	a
Diospyros littorea	1	a
Barringtonia racemosa	1	a
Acrostichum aureum	1	t
Acrostichum speciosum	1	t
Aegialitis annulata	1	t
Aegiceras corniculatum	1	t
Avicennia marina subsp. eucalyptifolia	1	t
Bruguiera cylindrica	1	t
Bruguiera exaristata	1	t
Bruguiera gymnorhiza	1	t
Bruguiera parviflora	1	t
Bruguiera sexangula	1	t
Bruguiera x rhynchopetala	1	t
Camptostemon schultzii	1	t
Ceriops australis	1	t
Ceriops pseudodecandra	1	t
Ceriops tagal	1	t
Excoecaria agallocha subsp. agallocha	1	t
Lumnitzera littorea	1	t

Lumnitzera racemosa	1	t
Nypa fruticans	1	t
Osbornia octodonta	1	t
Pemphis acidula	1	t
Rhizophora apiculata	1	t
Rhizophora mucronata	1	t
Rhizophora stylosa	1	t
Rhizophora x larmackii	1	t
Scyphiphora hydrophylacea	1	t
Sonneratia alba	1	t
Sonneratia caseolaris	1	t
Sonneratia lanceolata	1	t
Sonneratia x gulngai	1	t
Xylocarpus granatum	1	t
Xylocarpus moluccensis	1	t
Australia wide: including hybrids	32	
Tomlinson recognises 67 true mangrove taxa globally	*82% of Aust. true mangroves in CYP* *48% of world true mangroves in CYP*	

References:

Bostock, P. D., & Holland, A. E. (2018). *Census of the Queensland Flora 2018*. Retrieved May 2019, from https://www.qld.gov.au/environment/plants-animals/plants/herbarium/flora-census

Appendix 6.5 Mapped ‘estuarine’ and ‘oceanic’ mangrove forest of CYP bioregion

Please see accompanying KML file, distributed with this document, showing the areas of mangrove forest mapped as ‘estuarine / deltaic’ and ‘oceanic / fringing’ across CYP bioregion.

Appendix 7 Signed statements of contribution

Signed statements of contributions.

Chapter	Details of publication on which chapter is based	Nature and extent of intellectual input of each author, including candidate	I confirm the author's contribution to this paper and consent to the inclusion of the paper in this thesis
2 and 4	Addicott, E., Newton, M., Laurance, S., Neldner, J., Laidlaw, M., & Butler, D. (2018). A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion. *Cunninghamia,* 18, 29 - 71.	Addicott conceived the idea, performed all analysis, chaired the peer-review workshop and wrote the paper. Newton provided intellectual assistance, helped with data collation and cleaning, assisted with running the workshop, and assisted with the production of tables and appendices. Laurance assisted with writing and editing. Neldner collected much of the data, provided the historical context of the work and edited the final manuscript. Laidlaw tested the new approach on other datasets within government, provided environmental datasets and analysis advice and commented on the final manuscript. Butler provided intellectual input to the workshop process and direction and edited the final manuscript.	Mark Newton Susan Laurance Melinda Laidlaw John Neldner Don Butler

Chapter	Details of publication on which chapter is based	Nature and extent of intellectual input of each author, including candidate	I confirm the author's contribution to this paper and consent to the inclusion of the paper in this thesis
3	Addicott, E., Laurance, S., Lyons, M., Butler, D., & Neldner, J. (2018). When rare species are not important: linking plot-based vegetation classifications and landscape-scale mapping in Australian savanna vegetation. *Community Ecology*, 19, 67-76. doi:10.1556/168.2018.19.1.7	Addicott conceived the idea, performed most of the analysis and wrote the paper. Laurance provided intellectual support and helped write the manuscript. Lyons provided the analysis using Generalised Linear Modelling and the AIC information metric and help with interpreting the results and editorial assistance. Butler provided intellectual support and comments on the manuscript. Neldner collected much of the original data and commented on the manuscript.	Susan Laurance Mitchell Lyons Don Butler John Neldner

Chapter	Details of publication on which chapter is based	Nature and extent of intellectual input of each author, including candidate	I confirm the author's contribution to this paper and consent to the inclusion of the paper in this thesis
5	Addicott, E., & Laurance, S. G. W. (2019). Supervised versus un-supervised classification: A quantitative comparison of plant communities in savanna vegetation. *Applied Vegetation Science,* 22, 373-382. doi:10.1111/avsc.12442	Addicott conceived the idea, ran all analyses and wrote the manuscript. Laurance provided intellectual assistance and helped write the paper.	Susan Laurance

Chapter	Details of publication on which chapter is based	Nature and extent of intellectual input of each author, including candidate	I confirm the author's contribution to this paper and consent to the inclusion of the paper in this thesis
6	Addicott, E., Laurance, S. G. W., Bannink, P. & Thompson, S. Inter-tidal plant communities in north-eastern Australia, their relative role in carbon sequestration and vulnerability to extreme climate events: Baseline conservation information, in review in the journal *Aquatic Conservation: Freshwater and Marine Ecosystems*	Addicott conceived the idea, ran analysis (except as mentioned below) and wrote the manuscript. Laurance provided intellectual assistance, analysis advice and edited the paper, Bannink ran GIS analysis and commented on the paper, Thompson provided information regarding non-mangrove dieback and commented on the manuscript.	Susan Laurance Peter Bannink 29/08/2019 Simon Thompson 26/08/2019